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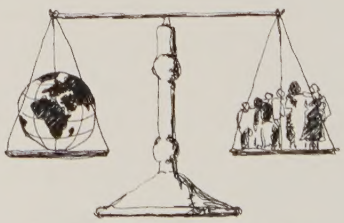

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# The grass is greener: Farmers' experiences with pastured poultry

Kathleen Hilimire

Department of Environmental Studies, University of California, 1156 High Street, Santa Cruz, CA 95064, USA.
Corresponding author: khilimir@ucsc.edu

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## Abstract

Rearing poultry outdoors on pasture for egg and meat production is a growing industry in the USA. However, little has been written on farmers' experiences with this practice. This paper presents the results of in-depth interviews and surveys with 18 California pastured poultry producers (62% response rate). Although the sample size of growers is small in absolute numbers, it represents 62% of nearly all pastured poultry growers in California, a state known for the pioneering work of growers who specialize in alternative agricultural practices. Such alternative systems merit critical analysis in order to refine their implementation and contribution to sustainable food production. This research aims to provide such critical analysis of pastured poultry production, a highly innovative and emergent practice. Growers were queried concerning the values that brought them to participate in alternative animal production, the benefits and challenges of integrating pastured poultry into farming operations, the use of heritage and industrial breeds, and management practices. Results showed that the influence of Joel Salatin, farmer and author of *Pastured Poultry Profits*, tied with farmer desire to sustainably produce meat and/or eggs as the main drivers respondents gave for raising pastured poultry (39% each). Farmers reported the primary benefit of pastured poultry was soil fertility (61%), followed by marketing appeal (44%). The most commonly cited challenge to pastured poultry growers was predation of birds (44%) followed by cost of feed (22%). Pastured poultry were directly profitable to 50% of farmers, although 78% of respondents cited indirect profits through savings on items such as fertilizer and pest management. This paper places these results in the context of value-laden farmer decision-making.

**Key words:** poultry, pasture, alternative food network, farmer, eggs, survey

## Introduction

Livestock sit at conflicting crossroads in the USA; one road is defined by ultra-efficient confined production and the other road reaches into an older history, pre-1920s, when animals were raised outdoors as a matter of course[1]. The second path re-emerges today in pasture-based livestock agriculture. At this uneasy crossroads, former vegetarians are doing the slaughtering and most sales transactions happen under the radar of agricultural statistics, involving no more than producer and consumer. Pastured livestock are a rapidly growing sector of alternative agriculture[2]. Portrayed as a form of meat and animal product production that allows natural animal behavior and positive environmental outcomes, pastured livestock enterprises contain 'embedded qualities'[3] of naturalness and environmental soundness at the core of their identities. In pastured animal production, the 'moral economy'[4] is emphasized over ultra-efficient production successes that define US industrial meat and animal production. This paper reports on one particular pastured livestock, poultry and growers' experiences with profitability, whole-farm management and ethical dilemmas in this emerging 'alternative agro-food network'[5].

### *Alternative poultry production*

Alternative meat production is one of the fastest-growing sectors of the US food system with organic poultry comprising 60% of this growth[6]. Twelve percent of consumers in the USA regularly purchase organic poultry and 15% buy organic eggs[7]. Poultry is considered a gateway food in the organic sector with consumers' first purchasing organic poultry and then other organic products[6,8]. Similar to organic poultry, pastured poultry are promoted as humanely raised, environmentally friendly and beneficial to food nutritional quality[2]. Pastured production differs from organic management, however, in that poultry live on and consume pasture, whereas organic regulations stipulate only that birds have access to the outdoors. Pastured poultry are raised outdoors and fed a grain-based diet that is naturally supplemented when birds graze their pasture[9].

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Studies regarding pastured poultry are few and primarily address meat and egg quality. For example, Ponte et al.[9] found that pasturing poultry significantly increased the *n*-3 (omega-3) fatty acid content of meat, whereas Karsten et al.[2] found higher vitamin A, vitamin E and *n*-3 fatty acid content in eggs from pastured poultry in comparison to caged birds with no access to pasture. These studies shed light on product outcomes in this new industry but do not assess grower outcomes of participation with a new alternative agricultural practice. Analysis of farmers' experiences with pastured poultry is critical to assessing areas for future research and to informing alternative agro-food network theory.

### *The farmer experience of alternative animal production*

Growers that raise pastured poultry are participants in a new management practice. Although poultry have been reared outdoors for centuries, the practice of raising poultry for commercial sale at scale without confinement is a new one[10]. The conventional poultry industry in the USA is characterized by high efficiency, confinement and extreme vertical integration[10]. Pastured poultry growers face steep price competition with the conventional industry and must rely on niche marketing[10]. In order to analyze growers' experiences with this new practice, the objectives of this research were: (i) to explain what drives farmers to participate in pastured poultry production, (ii) to describe the benefits and challenges farmers experience and (iii) to analyze the most contentious debates between grower participants in the emerging pastured poultry alternative agro-food network.

## Methods

This paper reports on the results of a survey that queried farmers on firsthand experiences with pastured poultry agriculture in California. Survey administration occurred between April and August 2010. All pastured poultry growers in California were sought for inclusion in the study, a small group given the newness of the practice (Fig. 1). Requests for participation were sent out to all poultry farmers in the California Eat Wild network (http://www.eatwild.com). Additional requests for participation were sent to all pastured poultry farmers found via the California Certified Organic Farmers (CCOF) database and by word of mouth. The response rate for the survey was 62% of 29 requests for participation. The survey was delivered via two channels, in-person administration or online completion. Thirty-nine percent of growers opted for in-person survey completion, whereas 61% completed the survey online.

The survey instrument addressed background information regarding location of farm, organic certification, quantity of farm animals and acreage of crops. Producers were asked to comment on the benefits and challenges of raising poultry as well as on specific management practices (e.g., managing predators and maintaining pasture). Farmers were then asked to comment on the profitability of pastured poultry and what stimulated them to raise pastured poultry. All questions were open ended allowing farmers to provide as much information as they preferred. A copy of the survey will be available at: http://www.surveymonkey.com/s/VDNWKMQ.

Survey responses were coded by topic ([11]: p. 400). For example, responses to the question on what inspired farmers to work with poultry were coded as 'exposure to Joel Salatin', 'profitability' and 'pest control'. Themes and relationships between coded variables were assessed quantitatively. For example, profitability was analyzed for a relationship to farm product (e.g., meat versus eggs). Data could not be analyzed by geographic region because using word of mouth to find survey participants necessarily skewed the geographic range of respondents to the Santa Cruz, California area, the location of the author's university. The following equations were used to calculate breed richness. Breed diversity was calculated as

$$R = C_a + C_b + C_{...}$$

where $R$ is breed richness and $C$ represents a count of 1 for each distinct poultry breed. The average breed richness across all farms was calculated as:

$$\overline{R} = (\sum R_1, R_2, R_{...})/F$$

where the sum of breed diversity at each farm was divided by $F$ for the total number of farms. Finally, each breed's relative abundance was determined as:

$$A = (N_c)/(\sum R_1, R_2, R_{...})$$

where $A$ is abundance, $N$ is the number of farms with a particular breed, $C$, and the denominator accounts for the total breed richness of all farms.

## Results

### *Farm information*

Eighteen pastured poultry farms participated in this study, all located in the state of California. Thirty-three percent of farms were certified organic, but 58% of non-certified farms specified that they used organic practices. All farms had layers; the average number of birds/year/farm was 591 (range = 50–3000; median = 300). Sixty-seven percent of farms also raised meat chickens; the average number of birds/year/farm was 5066 (range = 200–20,000; median = 1150).

### *Drivers of pastured poultry*

The influence of well-known farmer, Joel Salatin, tied with the desire of farmers to sustainably produce meat and/or eggs as the primary drivers California survey respondents gave for raising pastured poultry (Table 1). Following those

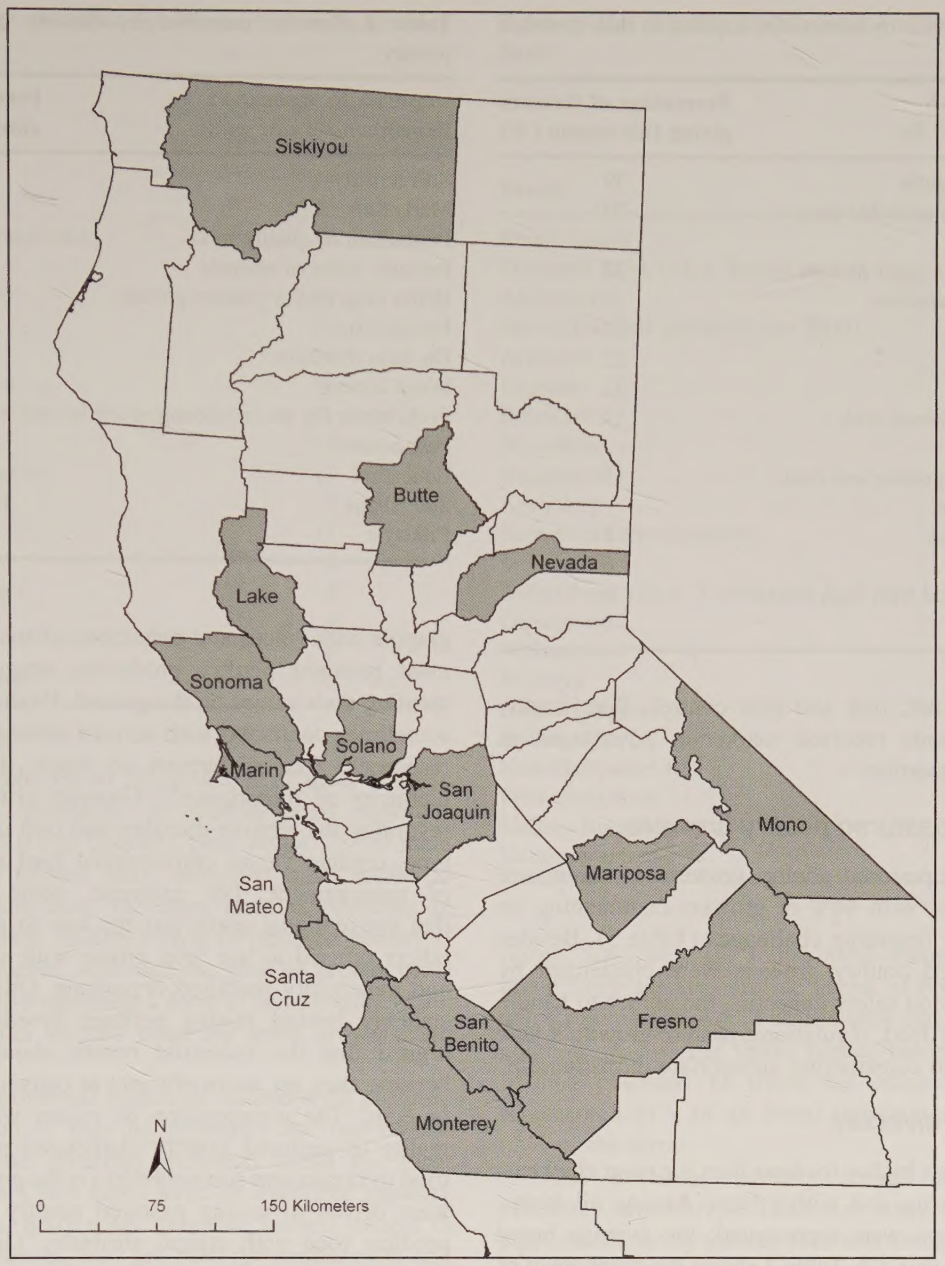


**Figure 1.** Map of California counties where the survey was administered.

two reasons were a suite of responses related to the role of poultry at the farm-scale, such as improving soil fertility with manure, birds cleaning up fruit and vegetable waste on farm, and chickens aiding in pest management through predation. Other farmers were motivated to raise pastured poultry because of perceived profitability.

## *Benefits of pastured poultry*

Farmers were asked what benefits they experienced from having poultry (Table 2). The top response was soil fertility, and approximately one-third of respondents went on to say that enhanced soil fertility contributed to better crop and/or pasture growth. Other common responses were with regard to marketing benefits and the production of quality food as a benefit. Many growers also commented on the intrinsic value of animals as a benefit.

## *Pastured poultry profits*

Growers responded with lengthy qualitative responses to the question, 'Has raising poultry been a profitable endeavor for you? Why or why not?' Fifty percent of farmers reported *direct* profitability from pastured poultry. There was no interaction between profitability and whether a farm had meat chickens versus layers or whether a farm was certified organic. In response to the same question, 78% of growers reported that pastured poultry were *indirectly* profitable because they attracted new customers to the farm, enhanced customer loyalty and contributed to

**Table 1.** Reasons given by farmers for deciding to raise pastured poultry.

| Reason | Percentage of farmers giving this reason (%) |
|---|---|
| Exposure to Joel Salatin | 39 |
| Desire to produce sustainable meat and/or eggs | 39 |
| Pasture poultry are a good fit with other farming operations | 22 |
| Profitability | 22 |
| Animal person | 22 |
| Soil fertility | 22 |
| Exposure to other farmer with pastured poultry | 17 |
| Poultry clean up vegetable and fruit waste on the farm | 11 |
| Educational purposes | 11 |
| Pest control | 6 |
| Desire to raise a food with high protein | 6 |
| Unique product | 6 |

**Table 2.** Benefits described by farmers of raising pastured poultry.

| Benefit | Percentage of farmers citing this benefit (%) |
|---|---|
| Soil fertility | 61 |
| Marketing | 44 |
| Production of quality food | 44 |
| Intrinsic value of animals | 39 |
| Better crop and/or pasture growth | 28 |
| Profitability | 28 |
| On-farm diversity | 22 |
| Weed control | 17 |
| Feels better for the environment | 17 |
| Pest control | 17 |
| Education | 11 |
| Soil tillage | 11 |
| Lifestyle | 6 |

savings on fertilizer, fuel and pest control. Specifically, 44% of respondents reported marketing advantages in response to this question.

### *Challenges to pastured poultry growers*

The top issue for pastured poultry growers was carnivore predation of birds, with 44% of growers commenting on this in a question regarding challenges (Table 3). Besides predators, pastured poultry growers were challenged by costs and labor, food safety concerns, the ability to handle large quantities of feed, regulations, pasture growth, a lack of background and constructing appropriate infrastructure.

### *Chicken breed diversity*

Breed diversity was higher for hens than for meat chickens, both across all farms and within farm. Among all farms, 21 breeds of layers were represented; the average breed richness per farm was 4.3. Table 4 shows the breakdown of richness by breed.

Far fewer breeds made up the diversity of meat chickens; six breeds were represented across all 12 farms surveyed with meat chickens. Of these, the industrial Cornish Cross represented 47% of all breed richness and was reared on 58% of the farms. Freedom Rangers (a.k.a. Colored Rangers or *Poulet Rouge*), a bird known for its livability on pasture, represented 27% of breed richness. K-22's, Plymouth Rock/Brit hybrid, Slow White Broilers and Slow Red Broilers were all reported by one farm each.

## Discussion

### *Farmer identity: the productivist meets the activist*

Just as the consumer faced with a refrigerated case of eggs bearing labels such as cage-free, organic and local must grapple with values and definitions of sustainability, so too must pastured poultry producers struggle when implementing their values on the ground. Production imperatives sometimes conflicted with activist strategies in the survey responses, forcing farmers to place their values in a hierarchy of importance[12]. Growers critiqued each other regarding the relative morality and cost of breed type, pen type, organic versus conventional feed and other aspects of management. For example, some farmers argued that conventional grain was the key to profitability while others refused to use feed grown with synthetic products and genetically modified organisms. Other growers stood squarely behind raising heritage breeds, whereas some argued that the industrial breeds should only be used because they are more efficient at converting feed to meat or eggs. The juxtaposition of values with the economic reality of pastured poultry challenged producers as they tried to implement sustainability on the ground, yet growers were driven to raising pastured poultry by the desire to produce food with ethical attributes. Growers were also driven by the influence of one particularly outspoken advocate of pastured poultry, Joel Salatin.

### *Why pastured poultry?*

Joel Salatin, a rancher from Virginia, leader in pastured meat chicken production and author of *Pastured Poultry Profits*[13], was a main driver for farmers to raise pastured poultry. He captured the public's imagination when he was featured in the bestseller, *Omnivore's Dilemma*, by food journalist Michael Pollan[14], and his impact on the swiftly growing pastured poultry movement cannot be underestimated. He emerged as an opinion leader in this survey, inspiring many of the survey respondents to enter into the new pastured poultry industry.

One of Salatin's most pivotal contributions to pastured poultry production has been a floorless moveable field pen, also called a chicken tractor. Moveable pens were

**Table 3.** Challenges described by farmers to raising pastured poultry.

| Challenges | Percentage of farmers citing this challenge (%) |
|---|---|
| Predators | 44 |
| Cost of feed | 22 |
| Too much work/cannot take vacations | 17 |
| Cost of organic feed | 17 |
| Not profitable | 11 |
| Cost of labor | 11 |
| Food safety concerns | 11 |
| Handling large quantities of feed | 11 |
| Regulations | 11 |
| Pasture would not grow | 11 |
| Lack of background | 11 |
| Infrastructure | 11 |
| Cost of insurance | 6 |
| Sales and marketing | 6 |
| Weather/climate | 6 |
| Disease | 6 |
| Lack of local breeders | 6 |

developed as a solution to the problem of meat chicken susceptibility to predators in pasture-based systems. Salatin's $3 \times 3.7 \times 0.6$ m floorless pens house 75–90 meat chickens and are moved daily, affording birds access to pasture, insects and dust, the three main components in natural bird behavior. Salatin's style of pastured poultry is controversial, however. Although the pens do protect birds on pasture, they also represent a form of semi-confinement, which is an animal welfare issue for some growers and consumers.

The other controversial aspect of Salatin-style pastured poultry production is that he promotes the use of the Cornish Cross, the principal industrial breed for meat chickens. Cornish Cross birds have an excellent capacity to convert feed to body mass, reaching harvest weight in 6–8 weeks[15]. However, they are bred for confinement and not pasture-based systems. Although they can survive and sometimes thrive in pasture-based systems, they may have higher mortality, greater incidence of heart disease and more leg problems than slower-growing varieties[16]. Despite the health issues of Cornish Cross birds, they can be a profitable choice for pasture and for confinement alike because of their grain-to-meat efficiency[16].

## *Chicken breed diversity*

The great efficiency of the Cornish Cross was reflected in the breed diversity results, which showed higher diversity for hens than for meat chickens, both across all farms and within farm. Among all farms, 21 breeds of layers were represented. Rhode Islands and Plymouth Rocks topped the list and are known to be prolific layers of brown eggs. Ameraucanas were third on the list; they are reliable layers of blue eggs, a trait that helps growers market a colorful

**Table 4.** Hen diversity and abundance of breed type among all farms.

| Breed | Abundance across all farms (as measured by % of total breed richness) |
|---|---|
| Rhode Island | 12 |
| Plymouth Rock (a.k.a. Barred rock) | 11 |
| Ameraucana | 11 |
| Star (a.k.a. Red and Black sex links) | 9 |
| Araucana | 8 |
| Leghorn | 8 |
| Delaware | 5 |
| Wyandotte | 5 |
| Andalusian | 3 |
| Australorp | 3 |
| Bantam (all true Bantams) | 3 |
| Marans | 3 |
| Naked Neck (a.k.a. Turken) | 3 |
| Orpington | 3 |
| Welsummer | 3 |
| Buckeye | 2 |
| Golden Nugget | 2 |
| Hyline | 2 |
| New Hampshire Red | 2 |
| Production Reds | 2 |
| Sicilian Buttercup | 2 |
| Unknown mixes | 2 |

dozen. Stars were the fourth most abundantly represented breed; they are also prolific layers of brown eggs and are valued by hen growers because they can be sexed by their color as chicks. Far fewer breeds made up the diversity of meat chickens. Of these, the industrial Cornish Cross represented 47% of all breed richness and was reared on 58% of the farms.

These disparate diversity results are explained by efficiency imperatives. There are many varieties of layers that can efficiently convert feed and forage to eggs, whereas Cornish Crosses are currently the most efficient meat chickens. As fast growers, they are less expensive to rear to market weight than slower-growing breeds[16,17]. This creates a paradox for pastured meat chicken producers. They may be endeavoring to satisfy animal welfare criteria, but economic imperatives push them toward selecting industrial genetic varieties with lower livability on pasture than slower-growing varieties. This paradox is inextricably tied to the chicken digestive track and the cost of feed. Chickens, unlike ruminants such as cows, do not fit comfortably into the grassfed world. Poultry do not have the digestive system of ruminants that allows them to digest fiber and convert grass into energy[18,19]. So although pasturing poultry does increase the nutritional content of meat and eggs[2,9] and affords access to protein from legume foraging and invertebrate predation, it does not reduce the amount of carbohydrates required by poultry, meaning that poultry growers must supplement pasture with grain-based

feeds, which can be expensive. In California, organic feed can cost nearly twice as much as conventional feed; typical conventional feed sells for $\$0.53\,kg^{-1}$ ($\$0.24\,lb^{-1}$), while organic feed costs $\$0.97\,kg^{-1}$ ($\$0.44\,lb^{-1}$). Growers in this alternative agro-food network, then, are caught in the space of justifying every choice as they solve for myriad unknowns, prioritized on values at times productivist and at times activist.

### *The grass is greener: indirect profits*

Although the top reason that growers cited for choosing to raise pastured poultry was the influence of Joel Salatin and his *Pastured Poultry Profits*[13] book, in this survey, only 50% of farmers reported *direct* profitability from pastured poultry. However, pastured poultry were *indirectly* profitable to 78% of surveyed growers because they attracted new customers to the farm, enhanced customer loyalty, and contributed to savings on fertilizer, fuel and pest control. One farmer said of indirect profits, 'With the price of the organic feed we usually come just over break even price. However if we factor in the value of what we'd have to pay for a fertility or bug control program then it still makes sense', and another responded, 'We make $6 per bird, but the fertilizer makes it worth it'. Growers reported both that chickens directly fertilize soil and that droppings are used as an ingredient in compost. '[We see an] improved topsoil ecosystem. [The] grass is darker green, grows taller, and soil retains moisture longer because of chickens, pasture, not spraying and not tilling'.

On a farm-finances scale, poultry were described as indirectly profitable because of their contribution to the overall marketability of a farm. Farmers said things like, '"[Having pastured poultry] builds customer loyalty because we have a product year-round", "Customers love ethical products", and "We stand out more [than crop-only farmers] at farmers' markets"'. To stand out in a direct marketing environment such as a farmers' market or CSA (Community Supported Agriculture), growers have to 'embed values' into the food they grow and sell[20], a process abetted by the advertisement of an ethical product. Consumers, especially in a direct-marketing environment, may be likely to align themselves with producers whom they perceive as having similar values. Animal products are also a unique item at farmers' markets, where fruits and vegetables are more common. One such grower said, 'You might even think of it as partly an advertising expense'.

Growers also cited the production of safe, high-quality food among the benefits of raising pastured poultry, saying things such as, 'We eat safe healthy and wholesome food, as do our customers'. Developing a niche in this setting can be critical to successful marketing[21]. In terms of overall profitability, then, the results of this study suggest that pastured poultry combine well with other farming activities and are best suited to diverse farms where the benefits of soil fertility and customer loyalty add value to other products.

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# Use of legume green manures as nitrogen sources for corn production

Matt Liebman[1]*, Rhonda L. Graef[2], Daniel Nettleton[3], and Cynthia A. Cambardella[4]

[1]Department of Agronomy, Iowa State University, Ames, IA 50011, USA.
[2]USDA-ARS, National Soil Erosion Research Laboratory, West Lafayette, IN 47907, USA.
[3]Department of Statistics, 2115 Snedecor Hall, Iowa State University, Ames, IA 50011, USA.
[4]USDA-ARS, National Laboratory for Agriculture and the Environment, Ames, IA 50011, USA.
*Corresponding author: mliebman@iastate.edu

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## Abstract

Recent volatility in supplies and prices of natural gas and synthetic nitrogen (N) fertilizer suggests a need to develop and refine alternative strategies for supplying N to corn. In this study, conducted in north-eastern Iowa, we examined the use of red clover and alfalfa green manures as means of supplying N to a succeeding corn crop. Red clover intercropped with oat produced significantly more biomass and contained more N than alfalfa intercropped with oat. Tilling green manures in the fall or delaying tillage until the following spring did not have a consistent effect on green manure N content. Without N fertilizer, corn grain yield following oat–red clover and oat–alfalfa was 25–63% greater than following oat grown alone, but at the highest fertilizer rate (202 kg N $ha^{-1}$), there was no difference in corn yield between oat–legume and oat-alone treatments. These patterns support the premise that legume green manure effects on corn yield were N-related. Red clover green manure had an N fertilizer replacement value for corn of 87–184 kg N $ha^{-1}$; alfalfa supplied corn with the equivalent of 70–121 kg N $ha^{-1}$. At a fossil energy cost for N fertilizer of 57 MJ $kg^{-1}$ N, reducing synthetic N fertilizer applications to corn by 70–184 kg N $ha^{-1}$ would represent a fossil fuel savings of 3990–10,488 MJ $ha^{-1}$, equivalent to the energy content of 104–274 $m^3$ of natural gas. These types of savings are likely to become increasingly important as fossil energy supplies become scarcer and fertilizer prices rise.

**Key words:** alfalfa, corn production systems, legume green manure, nitrogen fertilizer replacement value, red clover

## Introduction

Nitrogen (N) fertility is a critical component of corn (*Zea mays* L.) production. Within the USA, N is the nutrient supplied to crops in the greatest quantities as synthetic fertilizer and more N is applied to corn than any other crop[1]. In 2005, US farmers applied $4.6 \times 10^6$ Mg of synthetic N fertilizer to 96% of the area planted with corn at a mean rate of 155 kg N $ha^{-1}$.[1]

N fertilizer is not only applied to corn in large quantities, but it is also energy-intensive to synthesize. Consequently, it can comprise a major portion of the fossil-energy budget for corn production. Shapouri et al.[2] reported that N fertilizer requires, on average, 57 MJ of fossil energy (principally natural gas) per kg of N synthesized. Assuming an average fertilizer application rate of 155 kg N $ha^{-1}$ at an energy cost of 57 MJ $kg^{-1}$ N, and an energy content for natural gas of 38.3 MJ $m^{-3}$,[3] N application to 1 ha of corn would require the equivalent of 231 $m^3$ of natural gas. For conventional corn production in the Midwestern US, N fertilizer currently consumes 50% of the fossil energy used in the production cycle[2].

Recent volatility in supplies and prices of natural gas and synthetic N fertilizer[4] suggests a need to develop and refine alternative strategies for supplying N to corn that rely more on ecological processes, such as biological N fixation, than the Haber–Bosch industrial process. One such strategy is the use of legume green manures that fix atmospheric $N_2$ and are used as nutrient sources for subsequent crops[5]. Legume green manures suitable for the Midwestern US and southern Canada include alfalfa (*Medicago sativa* L.) and red clover (*Trifolium pratense* L.), which can be intercropped with small grains grown as cash or feed crops, preceding corn production[6–8].

N availability to a subsequent crop from legume green manures can vary with management options, such as tillage; soil and climate conditions, such as temperature and soil moisture; and tissue quality characteristics, such as

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**Table 1.** Chronology of field operations during 2000–2002.

| Field operation | Experiment 1 (2000–2001) | Experiment 2 (2001–2002) |
|---|---|---|
| Shallow cultivation | April 6, 2000 | April 16, 2001 |
| Oat, red clover and alfalfa planted | April 6, 2000 | April 17, 2001 |
| Oat harvested | July 21, 2000 | July 30, 2001 |
| Oat straw baled | July 21, 2000 | July 31, 2001 |
| Oat-only sub-plots mowed | August 11, September 1, 2000 | August 21, September 11, 2001 |
| Moldboard plowing for fall incorporation | November 14, 2000 | November 22, 2001 |
| Moldboard plowing for spring incorporation | April 20, 2001 | April 16, 2002 |
| Shallow cultivation | April 20, 2001 | April 17, 2002 |
| Shallow cultivation | May 12, 2001 | May 4, 2002 |
| Corn planted | May 12, 2001 | May 4, 2002 |
| Pre-emergence herbicide applied | May 12, 2001 | May 4, 2002 |
| Post-emergence herbicide applied | June 9, 2001 | June 1, 2002 |
| N fertilizer applied | June 11, 2001 | June 7, 2002 |
| Inter-row cultivation | June 11, 2001 | June 7, 2002 |
| Corn harvested | October 16, 2001 | October 6, 2002 |

content of C, N, cellulose, lignin and polyphenols[9–12]. In general, tillage induces more rapid mineralization of green manure materials via shredding and mixing with soil and associated micro-organisms; N release from green manures occurs slowly at or below 5°C, increases to a maximum at 30–35°C, and declines at higher temperatures; N release is favored by soil conditions that are neither exceptionally dry nor flooded (−0.1 to −1 MPa moisture tension); and tissues with carbon to nitrogen (C : N) ratios < 20 release N more quickly than those with C : N ratios >25[13,14].

N may be lost from incorporated green manures if between tillage and growth of a subsequent crop there is a long interval when soils are warm enough for organic matter mineralization and precipitation is great enough for nitrate leaching to occur[15]. Alternatively, if temperatures after incorporation of green manure are cool enough to inhibit mineralization, as might be the case with late fall plowing in temperate areas, nutrients might be retained until warmer conditions prevail in the spring. In much of the Midwestern US, it is often drier in the fall than it is in the spring, which facilitates plowing. However, fall plowing eliminates the erosion–reduction benefits of cover provided by an overwintering green manure crop.

Given the potential trade-offs and uncertainties associated with using and managing legume green manures, we conducted an experiment in north-eastern Iowa to determine the effects of using red clover or alfalfa as short-duration green manures preceding corn. Our specific objectives were to determine (i) biomass production and N-related characteristics of alfalfa and red clover intercropped with oat (*Avena sativa* L.) and incorporated in the fall or spring preceding corn production; (ii) corn yield responses to alfalfa and red clover green manure; and (iii) the N fertilizer replacement value (FRV) for corn of fall and spring tilled alfalfa and red clover green manures.

## Materials and Methods

### *Site description and experimental design*

Field research was conducted at the Iowa State University (ISU) Northeast Research and Demonstration Farm (NERDF) in Nashua, IA (42°57′N, 92°32′W). Predominant soil types are Kenyon loam (fine-loamy, mixed, mesic Typic Hapludolls), Floyd loam (fine-loamy, mixed, mesic Aquic Hapludolls) and Clyde loam (fine-loamy, mixed, mesic Typic Endoaquolls). Environmental data, including ambient air temperature, soil temperature and precipitation, were collected from the Iowa Environmental Mesonet[16] for the ISU-NERDF (Station ID: Nashua-a135879).

The first part of the field study (referred to as Experiment 1) was planted with oat and legumes in spring 2000 and was followed by field corn in 2001. The second part (referred to as Experiment 2) began with oat and legumes in spring 2001 and was followed by field corn in 2002. Oat and barley were grown in 1999 preceding the oat and legume crops of Experiment 1; soybean was grown in 2000 preceding the oat and legume crops of Experiment 2. Experiments 1 and 2 were conducted on separate fields, and each experiment was conducted using a randomized complete block, split-plot design. Whole-plot treatments comprised a factorial of three cropping systems (oat–alfalfa followed by corn, oat–red clover followed by corn and oat alone followed by corn) and two times of tillage (fall or spring plowing preceding corn). The six treatments were replicated four times and allocated randomly to strips within each of the two fields. The whole plots (61 m × 4.6 m) were split into randomized sub-plots (14.6 m × 4.6 m) that received a set of N fertilizer rates (0, 67, 134 and 202 kg N $ha^{-1}$) during the corn phase to determine the N FRV of the legume crops for the subsequent corn crop. Dates for field operations are presented in Table 1.

### Green manure phase

Oat and legume species were planted in April for each green-manure phase (Table 1). 'Jerry' oat was planted at a rate of 108 kg ha$^{-1}$, and 'Red Comet' red clover and 'Pioneer 53H81' alfalfa were planted at 17 kg ha$^{-1}$. Red clover and alfalfa seeds were treated with *Rhizobium* inoculants; alfalfa seeds were also treated with a fungicide (mefenoxam: (R)-2-[(2,6-dimethylphenyl)-methoxyacetyl-amino]-propionic acid methyl ester). Oat was harvested for grain and the straw was baled in July of both years. Oat plots were mowed 21 and 42 days after oat harvest to control weeds.

Prior to fall and spring incorporation, four 0.25 m$^2$ quadrats were taken in each green manure × tillage timing treatment replicate to determine biomass of legumes, oat and weeds. Plant material, both living and dead, was cut at a height of about 1 cm above ground level on October 17, 2000 and April 18, 2001 (Experiment 1) and October 17, 2001 and April 12, 2002 (Experiment 2), dried at 60°C for 48 h and then weighed. Biomass samples were processed through a 2.0-mm screen with a Thomas–Wiley mill and ground further with a Cyclone sample mill to pass through a 1.0-mm screen (Model 014, UDY Corporation, Fort Collins, CO, USA). Sub-samples were analyzed for total C and total N concentrations by dry combustion (modified Dumas method)[17] using a LECO CHN-2000 elemental analyzer (LECO Corp., St. Joseph, MI, USA).

Incorporation of oat residue, new oat growth from seeds not recovered by the combine, and legumes was done with a moldboard plowed to a depth of 25 cm in the fall and spring (Table 1). Immediately after spring incorporation, all plots were field cultivated once to break up the soil clods in the spring-incorporated plots and to control spring-emerging volunteer legumes in the fall-incorporated plots.

### Corn phase

Prior to corn planting, all plots were field cultivated twice for weed control and seedbed preparation. Corn was planted on May 12, 2001 (Golden Harvest 2390) and on May 4, 2002 (Golden Harvest 8250) at 80,000 seeds per hectare in 0.76 m rows. Pre-emergence herbicides were applied at the time of corn planting for grass weed control. In Experiment 1, acetochlor (2-chloro-2′-methyl-6′-ethyl-*N*-ethoxymethyl-acetanilide) was applied at a rate of 2.7 kg a.i. ha$^{-1}$; in Year 2, dimethenamid-P ([S]-2-chloro-N-[(1-methyl-2-methoxy) ethyl]-N-[2,4-dimethyl-thien-3-yl]-acetamide) was applied at a rate of 1.1 kg a.i. ha$^{-1}$. A post-emergence herbicide, dicamba (diglycolamine salt of 3,6-dichloro-o-anisic acid) was used, at rates of 280 g a.i. ha$^{-1}$ in Experiment 1 and 350 g a.i. ha$^{-1}$ in Experiment 2, to control broadleaf weeds.

Soil cores (3.2-cm-diameter) were taken at a depth of 15 cm during May and June in 2001 and 2002, respectively, at three or four randomly selected locations within each non-fertilized sub-plot. Soil samples were placed in sealed plastic bags and stored at 4°C prior to analysis. Field-moist samples were then pushed through an 8-mm-diameter sieve and soil water content was measured gravimetrically for sub-samples (15–20 g) after oven drying overnight at 105°C. Other field-moist 8-mm-sieved sub-samples (10 g) were extracted with 2 M KCl and $NO_3$-N concentration in the filtrate was determined using flow injection technology (Lachat Instruments, Milwaukee, WI, USA).

Urea fertilizer was applied at rates of 0, 67, 134 or 202 kg N ha$^{-1}$ at the vegetative sixth leaf corn-growth stage (V6)[18] using a drop spreader (Gandy Company, Owatonna, MN, USA) in the appropriate sub-plots on June 11, 2001 and June 7, 2002 (Table 1). The center holes of the spreader were sealed to prevent fertilizer from dropping into the corn whorl and damaging leaves. All plots were row cultivated once the same day to incorporate the fertilizer.

Corn plant density was determined on June 12, 2001 and June 21, 2002 by counting each plant in three central plot rows. Weed density was determined on May 29, 2001 and May 20, 2002 from eight randomly located 0.25 m$^2$ quadrats placed within the whole-plots. Cornstalk sections were taken on October 13, 2001 and September 27, 2002 from 15 randomly selected plants in three center rows of each sub-plot[19]. The samples were dried and ground and stalk $NO_3$-N concentration was quantified using flow-injection technology (Lachat Instruments, Milwaukee, WI, USA) on filtrate extracted from 0.250-g sub-samples with 50 ml 2 M KCl, shaken for 30 min. Three central sub-plot rows were harvested with a combine to determine grain yield; yields were adjusted to a moisture content of 155 g kg$^{-1}$.

### Statistical analyses

Data were analyzed using procedures in JMP 7.0.2[20]. Linear mixed-effects model analyses were conducted for plant variables in the green manure and corn phases of Experiments 1 and 2 using time of tillage and crop treatments as main plot factors, and N fertilizer rate treatments applied to corn as the split-plot factor. Random effects were included for replicate blocks and main plots. Fixed effects were included for tillage timing, crop system, fertilizer rate and all possible interactions among these factors. Mean separations among tillage timing × crop system combinations were conducted using linear contrasts. A repeated measures approach was used to examine the effects of time of tillage, preceding crop, sampling date and their interactions on soil $NO_3$-N concentrations in sub-plots not receiving N fertilizer within the corn phases of Experiments 1 and 2.

Corn grain yield responses to N fertilizer rate were further analyzed using polynomial regressions. Possible associations between corn population density and grain yield were examined by adding corn density as a covariate to models that included block, tillage timing, crop system, fertilizer rate and interactions among the factors. Relationships between soil $NO_3$-N concentrations, green manure characteristics and corn grain yield in sub-plots not

**Table 2.** Mean monthly precipitation and air temperature in 2000–2002 and 50-year averages (1951–2000), at the experimental sites.

| | Mean monthly rainfall | | | | Mean monthly air temperature | | | |
|---|---|---|---|---|---|---|---|---|
| **Month** | **2000** | **2001** | **2002** | **50-yr avg.** | **2000** | **2001** | **2002** | **50-yr avg.** |
| | mm | | | | °C | | | |
| January | 18 | 27 | 10 | 23 | −7.7 | −8.0 | −2.4 | −9.3 |
| February | 23 | 33 | 22 | 21 | −1.1 | −9.8 | −2.2 | −6.0 |
| March | 16 | 20 | 17 | 49 | 5.4 | −2.9 | −1.7 | 0.0 |
| April | 49 | 99 | 97 | 85 | 9.1 | 10.6 | 8.3 | 8.2 |
| May | 102 | 127 | 69 | 105 | 16.2 | 15.3 | 13.6 | 15.0 |
| June | 143 | 56 | 66 | 115 | 18.9 | 20.3 | 21.7 | 20.0 |
| July | 79 | 61 | 135 | 114 | 21.3 | 23.2 | 23.7 | 22.1 |
| August | 88 | 64 | 150 | 109 | 21.4 | 21.3 | 20.6 | 21.1 |
| September | 56 | 132 | 48 | 83 | 16.8 | 15.2 | 17.9 | 16.4 |
| October | 55 | 36 | 51 | 62 | 11.7 | 9.2 | 6.7 | 10.3 |
| November | 56 | 32 | 7 | 46 | −0.8 | 8.1 | 0.6 | 1.4 |
| December | 65 | 7 | 9 | 28 | −13.8 | −2.1 | −3.0 | −6.1 |

receiving N fertilizer were examined using correlation analyses.

The significance level for all comparisons and inclusion of terms in regression models was $P \leqslant 0.05$. Preliminary analyses indicated that experiment by treatment interactions occurred for many variables, so all data were analyzed separately by experiment.

## Results and Discussion

### *Environmental conditions*

**Air temperature.** Mean monthly air temperatures during the period of April–October for the green manure and corn phases of the experiment in Experiment 1 (2000–2001) and Experiment 2 (2001–2002) were similar to the 50-year average (Table 2). Air temperatures during the winter months (November–March) of the green manure phase averaged 7°C cooler in Experiment 1 compared to Experiment 2. Corn growing degree-days from planting to harvest were 1412 in Experiment 1 and 1533 in Experiment 2, with a base temperature of 10°C and an upper threshold of 30°C[21]. In both experiments, the growing degree-days accumulated were adequate for corn to achieve physiological maturity.

**Soil temperature.** Paul and Clark[13] noted that microbial decomposition of organic matter is slow below 5°C and above 40°C, and that the optimum range for nitrification is between 30 and 35°C. In temperate climates, nitrification rates are greatest in the spring through early fall and lowest in the winter. Soil temperatures recorded at a depth of 10 cm indicate that soil cooled to below 5°C by November 8, 2000 in Experiment 1 and by November 20, 2001 in Experiment 2 of the green-manure phase (data not shown). Fall incorporation of green-manure crops occurred within a week after these dates, so very little net N mineralization would have been likely at ambient temperatures until soil warmed the following spring. In the spring, soil temperatures maintained levels above 5°C by the first week in April in both experiments (data not shown). Spring incorporation of green manures occurred by the third week in April when soil temperatures were maintaining levels above 10°C. Assuming other conditions were adequate (moisture, aeration and organic matter composition), these soil temperatures would be warm enough to support net N mineralization for either the fall-incorporated biomass or the newly incorporated spring biomass. During the periods of corn production, air and soil temperature conditions should not have been limiting factors for microbial decomposition of organic matter.

**Precipitation.** In the green-manure phase of each crop sequence, total precipitation was greater in Experiment 1 (2000) than in Experiment 2 (2001) during the period of May–October (Table 2). In the months of June, July and August, mean monthly precipitation was 103 mm in the green-manure phase of Experiment 1 compared with 60 mm in Experiment 2. In the corn phase of the Experiment 1 crop sequence (2001), total precipitation was less than in Experiment 2 (2002) during the period from May to October, particularly in the months of June, July and August when the mean monthly rainfall was 60 mm in Year 1 compared with 117 mm in Experiment 2. Growing season totals for corn phases in both Experiment 1 (2001) and Experiment 2 (2002) were below the 50-year average (Table 2). However, neither drought nor flooded conditions were present in either year of corn production, suggesting that soil moisture availability was not an important constraint on green-manure decomposition.

### *Green manure phase*

**Green manure aboveground biomass.** In Experiment 1, mean biomass of spring tilled green manure treatments was 5660 kg DM $ha^{-1}$, which was significantly greater than the mean for fall tilled green manures (3490 kg $ha^{-1}$)

**Table 3.** Total aboveground biomass of crops and weeds, biomass N and C concentrations, total N and C contents and C : N for plant materials incorporated before corn production.

| Response | Fall tillage | | | Spring tillage | | | SE | Source of variation | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Oat–red clover | Oat–alfalfa | Oat alone | Oat–red clover | Oat–alfalfa | Oat alone | | Time of tillage (T) | Crop (C) | T × C |
| | | | | | | | | P>F | | |
| *Experiment 1*[1] | | | | | | | | | | |
| Biomass (kg ha$^{-1}$) | 5796 | 3121 | 1554 | 6845 | 5294 | 4840 | 735 | 0.0025 | 0.0023 | 0.3401 |
| N concentration (g N kg$^{-1}$) | 27 | 22 | 18 | 29 | 31 | 11 | 1 | 0.2785 | <0.0001 | <0.0001 |
| N content (kg N ha$^{-1}$) | 158 | 69 | 28 | 201 | 168 | 53 | 20 | 0.0043 | <0.0001 | 0.1944 |
| C concentration (g C kg$^{-1}$) | 423 | 445 | 407 | 422 | 367 | 359 | 19 | 0.0182 | 0.1572 | 0.1651 |
| C content (kg C ha$^{-1}$) | 2301 | 1306 | 594 | 2884 | 1870 | 1716 | 284 | 0.0205 | 0.0016 | 0.6011 |
| C:N ratio | 15 | 20 | 22 | 15 | 12 | 34 | 2 | 0.7920 | <0.0001 | 0.0019 |
| *Experiment 2* | | | | | | | | | | |
| Biomass (kg ha$^{-1}$) | 5721 | 2184 | 1254 | 4880 | 1667 | 2845 | 343 | 0.7848 | <0.0001 | 0.0058 |
| N concentration, (g N kg$^{-1}$) | 26 | 23 | 14 | 40 | 29 | 13 | 4 | 0.0485 | 0.0002 | 0.1303 |
| N content (kg N ha$^{-1}$) | 149 | 51 | 18 | 192 | 51 | 36 | 14 | 0.0867 | <0.0001 | 0.3092 |
| C concentration (g C kg$^{-1}$) | 429 | 436 | 407 | 381 | 378 | 325 | 9 | <0.0001 | 0.0005 | 0.1887 |
| C content (kg C ha$^{-1}$) | 2436 | 952 | 510 | 1833 | 630 | 918 | 125 | 0.1110 | <0.0001 | 0.0030 |
| C:N ratio | 17 | 19 | 30 | 10 | 15 | 26 | 2 | 0.0055 | <0.0001 | 0.6810 |

[1] Data for the fall tillage treatment are for samples collected on October 17, 2000 (Experiment 1) and October 17, 2001 (Experiment 2); data for the spring tillage treatment are for samples collected on April 18, 2001 (Experiment 1) and April 12, 2002 (Experiment 2).

(Table 3). Most of the biomass in the spring tillage oat-alone treatment was comprised of volunteer oat, whereas most of the biomass in the spring tillage oat–red clover and oat–alfalfa treatments was comprised of legume material, some of which was from new growth and some of which was residue of winter-killed fall growth (Fig. 1). Abundant precipitation in Experiment 1 (Table 2) may have increased soil moisture and contributed to increased late fall and early spring growth.

In Experiment 2, biomass of the spring tilled oat-alone treatment was significantly greater than the fall tilled oat-alone treatment, whereas tillage timing had no significant effect on biomass production in the oat–red clover and oat–alfalfa treatments (Table 3). We do not know why there was a reduced amount of legume biomass in the spring of Experiment 2 compared with Experiment 1. In particular, we did not monitor snow and ice cover, which might have affected winter kill, though soil temperatures did not differ greatly during the winter periods of Experiments 1 and 2 (data not shown).

In both experiments, the oat–red clover treatment produced significantly more biomass than the oat–alfalfa and the oat-alone treatments for both the fall and spring tillage regimes (Table 3). This was an expected result; in Wisconsin, when grown with a companion crop such as oat, red clover yield during the seeding year is generally 6–10 Mg ha$^{-1}$,[22] whereas alfalfa yield is 2–3 Mg ha$^{-1}$.[23]

**Botanical composition of aboveground biomass.** Aboveground biomass from the oat–red clover treatment consistently contained greater than 80% legumes, with a range of 83–94% (Fig. 1). Spring tilled biomass from the oat–red clover treatment tended to be slightly lower in legumes and higher in oat stubble and regrowth than the fall-tilled biomass; weeds were a very minor component. Winter kill or slower regrowth in the spring of red clover or volunteer oat growth in late fall or early spring may have been contributing factors to the lower legume percentage and increased oat percentage in the spring biomass.

The legume composition of the oat–alfalfa treatment biomass was similar in the fall for both experiments: 55% in Experiment 1 and 56% in Experiment 2 (Fig. 1). The percentage of legume material with spring tillage was 40–50% greater than with fall tillage for the oat–alfalfa treatment and similar to the legume concentration in the spring tilled oat–red clover treatment (Fig. 1).

After oat harvest, weeds were controlled in the oat-alone treatment by mowing twice after harvest (Table 1). Weed regrowth after mowing, however, was present in the fall biomass samples. Weeds were a major component in the fall tilled oat-alone treatment for Experiments 1 and 2, but not in the spring tilled oat-alone treatments (Fig. 1).

Weeds were minimal to non-existent in all the spring tillage treatments for both experiments (Fig. 1), presumably because the majority of weeds were annual grasses and had yet to germinate when biomass samples were taken.

**Aboveground biomass N content.** Green manure biomass N content was calculated from the product of aboveground biomass and N concentration. In both years, green manure N content was closely correlated with N concentration (Experiment 1: $r = 0.8022$, $p < 0.0001$; Experiment 2: $r = 0.8934$, $p < 0.0001$) (Table 3).

In Experiment 1, spring tilled treatments contained significantly more N than fall tilled treatments (141 versus

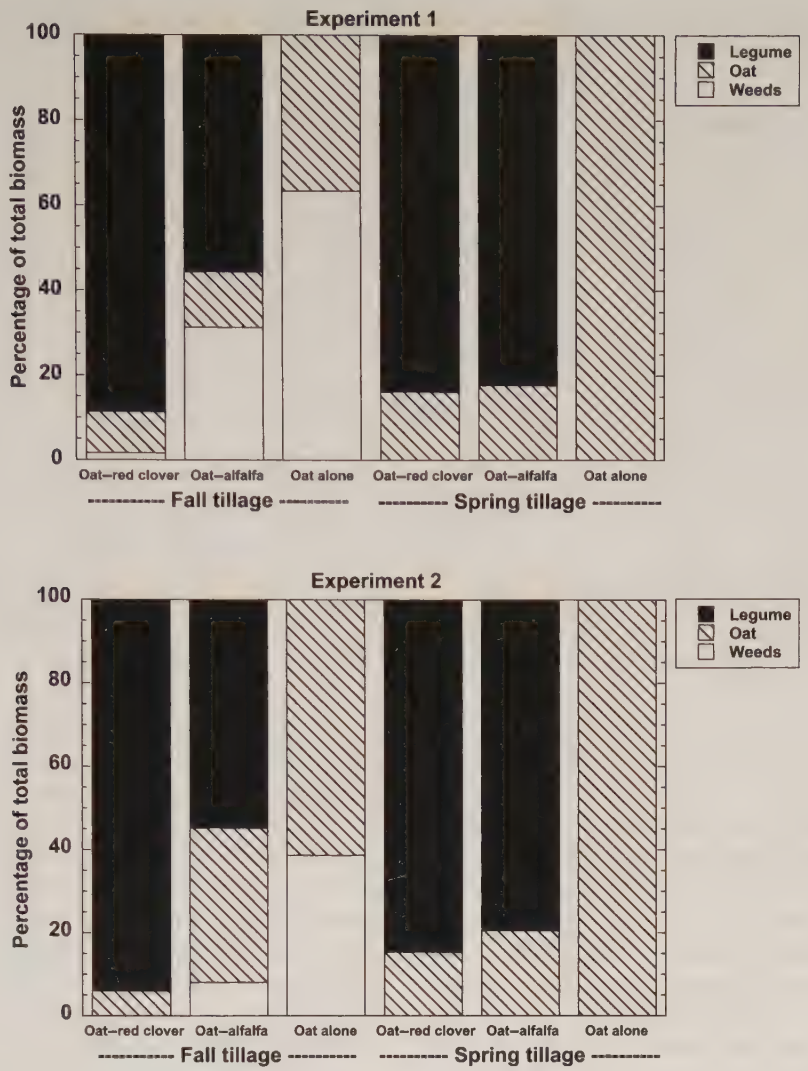


**Figure 1.** Botanical composition of green manure biomass.

85 kg N ha$^{-1}$; Table 3). Averaged over tillage regimes in Experiment 1, the oat–red clover treatment contained significantly more N than the oat–alfalfa treatment (179 versus 118 kg N ha$^{-1}$), and both of the oat–legume treatments contained significantly more N than the oat-alone treatment (40 kg N ha$^{-1}$) (Table 3).

In Experiment 2, tillage timing had no significant effect on green manure N content (Table 3), but crop effects were evident. Averaged across tillage regimes in Experiment 2, oat–red clover contained significantly more N (170 kg N ha$^{-1}$) than oat–alfalfa (51 kg N ha$^{-1}$) and oat-alone (27 kg N ha$^{-1}$), which did not differ statistically (Table 3).

**Aboveground biomass C:N ratio.** Soil environmental conditions and the composition of organic substrates are key factors affecting microbial decomposition of organic matter. The C:N ratio of organic materials is an important factor influencing net mineralization or immobilization of N[13,14]. In both years, C:N was strongly negatively correlated with green manure N concentration (Experiment 1: $r = -0.9410$, $P < 0.0001$; Experiment 2: $r = -0.9188$, $P < 0.0001$), but was not related to green manure C concentration (Experiment 1: $r = -0.0867$, $P = 0.7085$; Experiment 2: $r = -0.1114$, $P = 0.6043$) (Table 3).

In Experiment 1, green manure C:N was affected by an interaction between tillage timing and crop identity (Table 3). More specifically, C:N did not differ among crop types for the fall tillage regime, but was significantly greater in the oat-alone treatment than the oat–legume treatments for the spring tillage regime (Table 3).

In Experiment 2, fall-tilled treatments had significantly higher C:N values than did spring-tilled treatments (22 versus 17; Table 3). Averaged over tillage regimes, the oat-alone treatment had a significantly higher C:N value than did the oat–alfalfa treatment, which was in turn significantly higher than the oat–red clover treatment (28 versus 17 versus 13; Table 3).

With a C:N value close to or greater than 25 : 1 in the oat-alone treatments, there may have been immobilization of N and residual soil inorganic N may have been depleted[24]. In contrast, the lower C:N values for the oat-legume treatments favored more rapid microbial decomposition if other conditions were adequate.

**Table 4.** Densities of corn and weeds in corn as affected by time of tillage and preceding crop.

| Time of tillage and preceding crop | Corn[1] | | Weeds[2] | |
|---|---|---|---|---|
| | Experiment 1 | Experiment 2 | Experiment 1 | Experiment 2 |
| | no. $ha^{-1}$ | | no. $m^{-2}$ | |
| Fall tillage | | | | |
| Oat–red clover | 75,260 | 73,945 | 4 (1.8) | 7 (2.4) |
| Oat–alfalfa | 72,940 | 71,555 | 40 (5.9) | 25 (4.9) |
| Oat | 74,640 | 75,025 | 14 (3.7) | 27 (5.1) |
| Spring tillage | | | | |
| Oat–red clover | 74,330 | 62,815 | 5 (2.2) | 10 (2.9) |
| Oat–alfalfa | 71,550 | 63,300 | 32 (5.6) | 18 (4.1) |
| Oat | 73,560 | 72,320 | 17 (3.9) | 32 (5.4) |
| SE | 1560 | 1000 | (0.6) | (0.3) |
| *Source of variation* | | *P>F* | | |
| Time of tillage (T) | 0.1852 | <0.0001 | 0.8087 | 0.9944 |
| Preceding crop (C) | 0.0574 | 0.0023 | <0.0001 | <0.0001 |
| T×C | 0.9725 | 0.0472 | 0.8731 | 0.1240 |

[1] Corn densities were measured on June 12, 2001 (Experiment 1) and June 21, 2002 (Experiment 2).
[2] Weed densities were measured on May 29, 2001 (Experiment 1) and May 30, 2002 (Experiment 2) and were square root-transformed before analysis of variance. Means and standard errors of transformed data are shown in parentheses.

## *Corn phase*

**Corn density.** No significant differences in corn density were observed among treatments in Experiment 1, for which mean corn population density was 73,713 plants $ha^{-1}$ (Table 4). In Experiment 2, corn density in both the spring tillage oat–legume treatments was significantly lower than in the oat-alone treatment, and lower than in each of the fall tillage treatments (Table 4). Mean corn density for the two oat-legume treatments tilled in the spring was 63,058 plants $ha^{-1}$, whereas for the other crop × tillage timing treatments of Experiment 2, mean corn density was 73,211 plants $ha^{-1}$ (Table 4). Although we did not closely inspect corn seed to determine the cause of the lower corn density, reduction of corn density in spring-tilled oat–legume plots may have been related to damage by seed corn maggot (*Delia platura* (Meigen)). Hammond[25] reported that ovipositing female seedcorn maggots are attracted to freshly incorporated green plant matter and that damage to soybean seeds was greater following incorporation of a legume (alfalfa) than non-legumes (weeds and rye). Thus, spring incorporation of legumes in the present study may have favored this insect pest.

**Weed density in corn.** Weed density in corn was determined in late May after all plots had received a pre-emergence herbicide application at planting, but before post-emergence herbicide application (Table 1). Dominant weed species were yellow foxtail (*Setaria pumila* (Poir.) Roemer & J.A. Schultes), giant foxtail (*Setaria faberi* Herrm.), barnyardgrass (*Echinochloa crus-galli* (L.) Beauv.), common waterhemp (*Amaranthus rudis* Sauer), velvetleaf (*Abutilon theophrasti* Medik) and common lambsquarters (*Chenopodium album* L.).

For both experiments and both tillage timing treatments regimes, corn following oat–red clover had fewer weeds than did corn following oat–alfalfa and oat-alone (Table 4). This accords with the observation that in the fall, the percentage of total biomass comprised of weeds was less in the oat–red clover treatment than in the oat–alfalfa and oat-alone treatments (Fig. 1). Suppression of weed biomass production in the oat–red clover treatment was probably accompanied by suppression of weed seed production[26], resulting in fewer weeds in the subsequent corn crop. Mutch et al.[27] reported similar weed-suppressive effects for red clover. Weed density in corn was not significantly affected by tillage timing in either experiment (Table 4).

**Soil nitrate dynamics.** Repeated measures analyses indicated that soil $NO_3$-N concentration in corn was affected by significant interactions between sampling date and time of tillage and crop system factors. In general, $NO_3$-N concentrations were significantly higher following oat–red clover and oat–alfalfa than oat alone, and were affected inconsistently by the time of tillage (Table 5). The oat–red clover and oat–alfalfa treatments tended to show higher $NO_3$-N concentrations in May and early June before declining in late June, whereas $NO_3$-N concentrations in the oat-alone treatment did not vary significantly among sampling dates (Table 5).

Corn N uptake and growth are generally rapid in mid to late June in north-eastern Iowa and soil N availability during this period can strongly influence corn yield. Morris et al.[28] reported that for corn following alfalfa, late spring soil $NO_3$-N concentrations at or above 14 $\mu g\ g^{-1}$ in the top 30 cm of soil indicated that corn yield would be unlikely to respond to supplemental N fertilizer. Data from the present experiment represent $NO_3$-N concentrations in the top 15 cm of soil, so direct comparisons with Morris et al.'s

**Table 5.** Surface soil (0–15 cm) $NO_3$-N concentration at selected sampling dates in plots not receiving N fertilizer during the corn phase of Experiments 1 and 2.

| Sampling date | Fall tillage | | | Spring tillage | | | SE | Source of variation | | | Correlation with corn grain yield, Mg ha$^{-1}$ |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | Oat–red clover | Oat–alfalfa | Oat alone | Oat–red clover | Oat–alfalfa | Oat alone | | Time of tillage (T) | Crop (C) | T × C | |
| | ----------$\mu$g $NO_3$-N g$^{-1}$ soil---------- | | | | | | | ---------P>F--------- | | | |
| *Experiment 1*[1] | | | | | | | | | | | |
| May 14, 2001 | 10.63 a | 8.64 a | 6.48 a | 11.77 a | 11.54 a | 7.89 a | 1.42 | 0.1201 | 0.0170 | 0.8003 | 0.58** |
| May 29, 2001 | 15.62 b | 9.59 a | 4.32 a | 9.21 a | 10.44 a | 6.49 a | 1.80 | 0.4415 | 0.0007 | 0.0410 | 0.43* |
| June 12, 2001 | 10.04 a | 14.62 a | 8.48 a | 11.39 a | 13.58 a | 7.90 a | 1.84 | 0.9543 | 0.0193 | 0.7919 | 0.51* |
| June 26, 2001 | 7.42 b | 7.00 b | 3.73 a | 10.71 a | 5.66 b | 4.76 a | 1.52 | 0.4330 | 0.0207 | 0.3383 | 0.72*** |
| *Experiment 2* | | | | | | | | | | | |
| May 9, 2002 | 13.69 a | 11.96 a | 5.25 a | 12.61 b | 10.34 b | 6.64 a | 1.40 | 0.7021 | <0.0001 | 0.5206 | 0.72*** |
| May 23, 2002 | 12.19 b | 13.02 a | 4.83 a | 23.91 a | 14.78 a | 7.08 a | 2.37 | 0.0082 | <0.0001 | 0.0662 | 0.49* |
| June 7, 2002 | 12.00 b | 13.89 a | 8.01 a | 10.67 b | 16.65 a | 7.11 a | 1.79 | 0.9023 | 0.0003 | 0.4563 | 0.67*** |
| June 21, 2002 | 7.33 c | 8.07 b | 5.32 a | 16.57 a | 10.30 b | 4.24 b | 1.35 | 0.0026 | <0.0001 | 0.0011 | 0.45* |

[1] For each experiment, means within columns not followed by the same letter are significantly different at $P \leq 0.05$.
* $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$.

study are not possible. Nonetheless, it is notable that soil $NO_3$-N concentrations were above Morris et al.'s 14 $\mu$g g$^{-1}$ threshold in only four of the eight time of tillage × legume green manure × experiment combinations we observed.

**Corn stalk nitrate concentration.** Binford et al.[29] examined the relationship between corn grain yield and stalk $NO_3$-N concentration at plant maturity and determined that plants with <700 mg $NO_3$-N kg$^{-1}$ were N-deficient, those in the range of 700–2000 mg $NO_3$-N kg$^{-1}$ had sufficient amounts of N to reach their yield potential, and those with >2000 mg $NO_3$-N kg$^{-1}$ had received excessive amounts of N. In both Experiments 1 and 2 of the present study, fall tilled and spring tilled oat-alone treatments required the highest fertilizer rate used in the experiment (202 kg N ha$^{-1}$) to reach stalk concentrations above 700 mg $NO_3$-N kg$^{-1}$ (Table 6). In contrast, corn stalk analyses indicated optimum levels were reached in Experiment 1 at the 134 kg N ha$^{-1}$ fertilizer rate for fall tilled and spring tilled oat–red clover and oat–alfalfa treatments; in Experiment 2, optimum levels were attained for the oat–red clover and oat–alfalfa treatments at 67 kg N ha$^{-1}$ with fall tillage, and at 67 and 134 kg N ha$^{-1}$, respectively, with spring tillage (Table 6). These results indicate that treatments with legume green manure reached optimum corn stalk $NO_3$-N levels with less fertilizer than the oat-alone treatment.

Results of contrast analyses indicated that in Experiment 1, averaged over fertilizer and tillage treatments, both legume treatments produced higher stalk $NO_3$-N concentrations than the oat-alone treatment ($P < 0.0001$), but did not differ from each other ($P = 0.4405$). In Experiment 2, the red clover treatment produced higher stalk $NO_3$-N concentrations than the alfalfa treatment ($P = 0.0224$), and both legume treatments produced higher levels than the oat-alone treatment ($P < 0.0001$). Time of tillage did not significantly affect stalk $NO_3$-N concentrations in either experiment (Table 6).

**Corn yield.** Corn yield in Experiment 1 was significantly affected by an interaction between preceding crop and N fertilizer rate, but there were no main or interaction effects of tillage timing (Table 7). In contrast, corn yield in Experiment 2 was affected by a three-way interaction of preceding crop × tillage timing × N fertilizer rate (Table 7). In general, corn yields following red clover and alfalfa were similar, but greater than corn yields following oat-alone at lower fertilizer rates.

Regression analyses were used to further investigate the impacts of different preceding crops and times of tillage on corn yield response to N fertilizer. Mean, linear and quadratic models were fit for corn yield responses to varying N fertilizer rates for each preceding crop in Experiment 1, when there were no main or interactive effects of tillage timing, and each tillage timing × preceding crop combination in Experiment 2, when tillage timing effects were significant (Table 7). To make parsimonious prediction equations, we rejected model terms that were not significant (i.e., $P > 0.05$). In the case of corn following red clover in both the fall and spring tillage treatments of Experiment 2, there were no significant linear or quadratic trends, so we used mean responses across fertilizer rates. We also fit segmented regressions using quadratic with plateau models, as recommended by Cerrato and Blackmer[30], but found that simpler, non-segmented polynomial models fit the data better.

In both Experiments 1 and 2, corn responded considerably more strongly to N fertilizer in the oat-alone treatment than in the oat–red clover and oat–alfalfa treatments. This is evident in the lower intercept values and larger linear coefficients in the prediction equations for the oat-alone treatment as compared with the legume treatments

**Table 6.** Cornstalk $NO_3$-N concentration at maturity as affected by time of tillage, preceding crop and N fertilizer rate.

| Time of tillage | Preceding crop | Experiment 1[1] Fertilizer rate (kg N ha$^{-1}$) 0 | 67 | 134 | 202 | Experiment 2 Fertilizer rate (kg N ha$^{-1}$) 0 | 67 | 134 | 202 |
|---|---|---|---|---|---|---|---|---|---|
| | | mg $NO_3$-N kg$^{-1}$ | | | | | | | |
| Fall | Oat–red clover | 11 (2.0) | 548 (6.1) | 1002 (6.8) | 2991 (8.0) | 322 (5.4) | 1956 (7.5) | 3537 (8.2) | 4550 (8.4) |
| Fall | Oat–alfalfa | 14 (2.6) | 177 (5.0) | 1226 (7.1) | 2502 (7.8) | 437 (2.7) | 1182 (6.4) | 2330 (7.7) | 3042 (8.0) |
| Fall | Oat alone | 7 (1.0) | 9 (1.7) | 85 (4.1) | 921 (6.7) | 25 (1.9) | 41 (2.2) | 318 (5.4) | 1040 (6.8) |
| Spring | Oat–red clover | 91 (3.7) | 91 (3.7) | 1301 (7.1) | 2603 (7.9) | 108 (4.4) | 701 (6.5) | 2372 (7.7) | 3202 (8.0) |
| Spring | Oat–alfalfa | 13 (2.5) | 141 (4.8) | 1134 (6.9) | 2300 (7.7) | 39 (3.6) | 575 (6.0) | 1831 (7.4) | 2976 (8.0) |
| Spring | Oat alone | 5 (0.6) | 15 (2.6) | 33 (3.2) | 700 (6.5) | 50 (3.6) | 36 (3.6) | 137 (4.5) | 816 (6.6) |
| SE | | (1.2) | | | | (1.4) | | | |
| *Source of variation* | | *P>F* | | | | | | | |
| Time of tillage (T) | | 0.4834 | | | | 0.7824 | | | |
| Preceding crop (C) | | <0.0001 | | | | <0.0001 | | | |
| T × C | | 0.9796 | | | | 0.2017 | | | |
| Fertilizer rate (F) | | <0.0001 | | | | <0.0001 | | | |
| F × T | | 0.4836 | | | | 0.2562 | | | |
| F × C | | 0.7815 | | | | 0.2927 | | | |
| F × T × C | | 0.8343 | | | | 0.2425 | | | |

[1] Samples were collected on October 13, 2001 (Year 1) and September 27, 2002 (Year 2). Data were ln-transformed before analysis of variance. Means and standard errors of transformed data are shown in parentheses.

**Table 7.** Corn yield (at 155 g $H_2O$ kg$^{-1}$) as affected by time of tillage, preceding crop and N fertilizer rate.

| Time of tillage | Preceding crop | Experiment 1 Fertilizer rate, kg N ha$^{-1}$ 0 | 67 | 134 | 202 | Experiment 2 Fertilizer rate, kg N ha$^{-1}$ 0 | 67 | 134 | 202 |
|---|---|---|---|---|---|---|---|---|---|
| | | Mg ha$^{-1}$ | | | | | | | |
| Fall | Oat–red clover | 9.97 | 12.38 | 11.80 | 12.72 | 12.80 | 13.90 | 13.34 | 13.03 |
| Fall | Oat–alfalfa | 10.29 | 12.11 | 11.75 | 12.43 | 11.86 | 12.84 | 13.08 | 13.17 |
| Fall | Oat alone | 7.89 | 10.87 | 11.63 | 12.09 | 9.52 | 11.53 | 12.59 | 13.33 |
| Spring | Oat–red clover | 11.03 | 11.61 | 12.03 | 12.52 | 11.42 | 11.70 | 12.58 | 12.41 |
| Spring | Oat–alfalfa | 10.34 | 12.08 | 12.83 | 12.23 | 11.47 | 12.00 | 11.75 | 12.36 |
| Spring | Oat alone | 7.86 | 10.44 | 11.01 | 12.09 | 7.03 | 10.58 | 11.90 | 12.07 |
| SE | | 0.28 | | | | 0.34 | | | |
| *Source of variation* | | *P>F* | | | | | | | |
| Time of tillage (T) | | 0.9234 | | | | <0.0001 | | | |
| Preceding crop (C) | | <0.0001 | | | | <0.0001 | | | |
| T × C | | 0.2086 | | | | 0.4492 | | | |
| Fertilizer rate (F) | | <0.0001 | | | | <0.0001 | | | |
| F × T | | 0.0867 | | | | 0.3524 | | | |
| F × C | | <0.0001 | | | | <0.0001 | | | |
| F × T × C | | 0.0712 | | | | 0.0204 | | | |

(Table 8). The three-way interaction detected in Experiment 2 was the result of a stronger response to N fertilizer in the oat-alone treatment for spring tillage than for the fall tillage (Table 8).

Analysis of covariance indicated that corn population density had no significant association with corn yield in Experiment 1 ($P = 0.9843$), but did have a significant association with yield in Experiment 2 ($P = 0.0014$). More specifically, reductions in corn density that occurred with the use of legume green manures and spring tillage in Experiment 2 (Table 4) were significantly associated with lower corn yield.

In both experiments, corn yield in the 0 kg N ha$^{-1}$ fertilizer treatment was significantly positively correlated with green manure aboveground biomass (Experiment 1: $r = 0.55$, $P = 0.0051$; Experiment 2: $r = 0.43$, $P = 0.0377$),

**Table 8.** Prediction equations for corn yield response ($Mg\,ha^{-1}$) to N fertilizer rate ($kg\,N\,ha^{-1}$) for time of tillage × preceding crop combinations in the two experiments, and estimated fertilizer N equivalence of red clover and alfalfa. Standard errors are shown in parentheses.

| Experiment | Time of tillage | Preceding crop | Intercept | Linear coefficient | Quadratic coefficient | N fertilizer equivalent ($kg\,N\,ha^{-1}$) |
|---|---|---|---|---|---|---|
| 1 | Fall, spring | Oat–red clover | 10.8140 (0.2303) | 0.0094 (0.0018) | | 87 |
| 1 | Fall, spring | Oat–alfalfa | 10.3843 (0.1861) | 0.0287 (0.0044) | −0.000096 (0.000021) | 70 |
| 1 | Fall, spring | Oat alone | 7.9857 (0.2136) | 0.0422 (0.0051) | −0.000111 (0.000024) | |
| 2 | Fall | Oat–red clover | 13.2694 (0.1999) | | | 184 |
| 2 | Fall | Oat–alfalfa | 12.1081 (0.2735) | 0.0062 (0.0022) | | 121 |
| 2 | Fall | Oat alone | 9.8723 (0.4219) | 0.0185 (0.0034) | | |
| 2 | Spring | Oat–red clover | 12.0252 (0.2758) | | | 131 |
| 2 | Spring | Oat–alfalfa | 11.5368 (0.2100) | 0.0036 (0.0017) | | 104 |
| 2 | Spring | Oat alone | 7.0817 (0.5142) | 0.0622 (0.0123) | −0.000187 (0.000058) | |

but negatively correlated with green manure C:N ratio (Experiment 1: $r = -0.75$, $P < 0.0001$; Experiment 2: $r = -0.59$, $P < 0.0024$). Corn yield in the $0\,kg\,N\,ha^{-1}$ fertilizer treatment was also strongly and positively correlated with the green manure N concentration (Experiment 1: $r = 0.76$, $P < 0.0001$; Experiment 2: $r = 0.50$, $P = 0.0125$), green manure N content (Experiment 1: $r = 0.74$, $P < 0.0001$; Experiment 2: $r = 0.51$, $P = 0.0101$), and soil $NO_3$-N concentrations (Table 5). Taken together, these results indicate that legume green manures that produced the greatest amount of N-rich biomass contributed most strongly to higher corn yields under N limited conditions.

**Nitrogen FRV of legume green manures.** Legume green manures may have positive impacts on the yield of succeeding crops due to both N-related and non-N-related effects. As described by Pierce and Rice[31], a *non-N rotation effect* is the difference in yield of a target crop at a high fertilizer rate following legume or non-legume, whereas a *total rotation effect* is the yield difference observed following legume and non-legume crops when no fertilizer is applied. The difference between the *total rotation effect* and the *non-N effect* is the *N-rotation effect*.

When we compared the difference in corn yield between the oat–red clover and oat–alfalfa treatments versus the oat-alone treatment for each experiment and for each time of tillage, the yield difference at the $0\,kg\,N\,ha^{-1}$ fertilizer rate was significant in each case, whereas the yield difference at the $202\,kg\,N\,ha^{-1}$ fertilizer rate was not (Table 7). This indicated that the observed responses to legumes were N rather than non-N effects. Consequently, we used the regression equations for corn responses to N fertilizer (Table 8) to calculate N FRVs following a procedure described by Hesterman et al.[32]. Using the prediction equations for the oat-alone treatments, we calculated N FRVs as the amount of fertilizer needed to give corn yields equivalent to the *y*-intercepts in the prediction equations for the legume treatments.

FRVs for red clover ranged from 87 to $184\,kg\,N\,ha^{-1}$, whereas values for alfalfa ranged from 70 to $121\,kg\,N\,ha^{-1}$ (Table 8). We found no consistent patterns for tillage timing effects, but in each experiment, FRVs for red clover were higher than for alfalfa. Values calculated in the present study are similar to those reported by other researchers studying corn yield responses to short-duration legume green manures in temperate areas[6,7,33].

**Economic and energetic considerations.** The cost of synthetic N fertilizer rose more than 200% during the period of 1991–2006 and closely paralleled increases in the cost of natural gas[34], a key component of its manufacture. If it is assumed that over the longer term, supplies of natural gas will shrink and its cost will rise, then reducing reliance on synthetic N sources is likely to be a useful strategy for limiting costs of corn production. At a fossil energy cost of $57\,MJ\,kg^{-1}\,N$, reducing synthetic N fertilizer applications to corn by the FRVs calculated for legumes used in the present study—70–$184\,kg\,N\,ha^{-1}$—would represent a fossil fuel savings of 3990–10,488 MJ $ha^{-1}$, equivalent to the energy content of 104–$274\,m^3$ of natural gas[2,3].

Two factors are likely to favor the use of red clover over alfalfa as a short-duration green manure: its generally greater N FRV (Table 7) and its lower seed cost. An informal survey of seed suppliers in Iowa in 2011 indicated that red clover seed cost \$3.75–$6.10\,kg^{-1}$, depending on variety, whereas alfalfa seed cost \$7.10–$9.50\,kg^{-1}$. Thus, for the seeding rate used in this study for both species ($17\,kg\,ha^{-1}$), red clover would be a more economical choice for supplying N to corn. Assuming that (i) red clover seed would cost $\$4.50\,kg^{-1}$ and allow a reduction of fertilizer application to corn of $120\,kg\,N\,ha^{-1}$; (ii) alfalfa seed would cost $\$8.00\,kg^{-1}$ and allow a reduction in fertilizer application to corn of $90\,kg\,N\,ha^{-1}$; (iii) oat yield would be unaffected by the presence of legume intercrops[35]; and (iv) both red clover and alfalfa would be planted at $17\,kg\,ha^{-1}$; then the cost of N fertilizer at which planting legumes with oat would be economically competitive with planting oat alone would be $\$0.64\,kg^{-1}$ of N for red clover and $\$1.51\,kg^{-1}$ of N for alfalfa. As a point of reference, during the period of 2001–2010, N fertilizer

costs in Iowa varied from \$0.44 to \$1.50 $kg^{-1}$ of N, with a mean cost of \$0.73 $kg^{-1}$ of N[36].

The economics of a crop rotation system depend on yields, prices obtained for crop products, government subsidies, and input, land and labor costs. Relatively few studies have evaluated the economic performance of substituting or adding small grains and forage legumes to the corn–soybean system that dominates the US Midwest. Using data from two long-term cropping systems trials in Wisconsin, Chavas et al.[37] found that without government subsidy payments and organic premiums, a corn–soybean system receiving conventional chemical inputs was more profitable than a corn–soybean–wheat/red clover system under organic management; with subsidies and organic premiums, the more diverse system was more profitable. Cruse et al.[38] used 6 years of data from a cropping systems trial in Iowa to compare the economic performance of a conventionally managed corn–soybean system with that of a corn–soybean–small grain/red clover system that received reduced amounts of synthetic N fertilizer and herbicides, but some manure. The small grain crop was initially triticale and then oat. With the assumptions that (i) the manure was a waste product that incurred costs only for labor and machinery associated with its application, and (ii) no government subsidies or premiums were received for the crops, returns from the two systems were found to be statistically equivalent, differing by \$9 $ha^{-1}$ in favor of the corn–soybean system. However, fossil energy use, including energy used for producing N fertilizer, was 41% lower in the more diverse system[38], suggesting that increases in fossil fuel costs that were disproportionately greater than potential increases in crop value would make the corn–soybean–small grain/red clover economically advantageous. As noted by Cherr et al.[5], future research efforts concerning the use of green manure legumes should take a systemic approach toward identifying the full set of services that they might provide and the factors affecting their adoption by farmers, including economic and energetic efficiency.

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# A capacity assessment of New England's large animal slaughter facilities as relative to meat production for the regional food system

Chelsea Bardot Lewis[1]* and Christian J. Peters[2]

[1]Vermont Agency of Agriculture, Food, and Markets, Montpelier, VT 05620, USA.
[2]Friedman School of Nutrition Science and Policy, Tufts University, Boston, MA 02111, USA.
*Corresponding author: chelsea.lewis@state.vt.us

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## Abstract

Demand for locally and regionally produced meat has stimulated increased interest in livestock production among New England farmers. The region's livestock producers lament lack of access to slaughter and processing infrastructure. However, there is very little research on New England's slaughter industry to document this perceived problem. For this reason, we tested the hypothesis that a shortage of slaughter and processing infrastructure constrains the production of livestock for meat in New England. The region's large animal slaughter facility owners and managers were surveyed to determine current slaughter and processing capacity and identify challenges facilities face in meeting increased producer demand. The estimates of current capacity were then compared to USDA data on livestock slaughter and large animal marketings. The region's existing abattoirs could slaughter 63–84% of all animals marketed, but could process only 29–43%. New England's infrastructure for slaughter operated at only 38% of total physical capacity in 2009, while on-site processing infrastructure operated at 66% of total physical capacity (78% if only on-site inspected capacity is considered). Moreover, surveys with facility operators showed that the primary constraints faced by existing slaughterhouses are a shortage of skilled labor and the seasonality of the livestock industry, with periods of very high demand for slaughter in the fall and very low demand in the spring and early summer. Additional infrastructure, particularly for processing, would be needed were regional livestock production to increase. However, simply increasing physical capacity will not address the issues of labor availability and demand seasonality expressed by slaughter facility owners.

**Key words:** local food systems, regional food systems, livestock production, livestock processing, infrastructure

## Introduction

The popularity of the local foods movement has increased the demand for New England-produced meat products[1], as a range of concerns with the industrial model of meat production have been publicized[2–4]. Consumers are increasingly interested in higher-quality food, and include factors such as taste, nutritive quality, safety requirements, environmental protection, fair labor practices and animal welfare in their consumption decisions[5,6]. While the actual benefits of local foods on reducing greenhouse gas emissions and energy use are still a matter of debate[7], the increase in consumer demand for local is clear. For instance, the American Culinary Federation reported that locally sourced meat was the number two hot trend for 2010, just behind locally grown produce[8]. New England consumers have been especially enthusiastic about local foods. Vermont has the highest level of per-capita direct-to-consumer sales in the nation, with the rest of the New England states (Connecticut, Maine, Massachusetts, New Hampshire and Rhode Island) all among the top 15 nationally[9].

The region's producers have been eager to meet this increased demand, particularly as the changing structure of the dairy sector over the past 20 years[10] has led many farmers to explore alternative enterprises. Meat animal production is a common transitional enterprise, as New England dairy farms often have the land, equipment

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and infrastructure required to grow and store feed. Regional, small-scale livestock production is perceived to offer a range of benefits to farmers, including ecological sustainability, improved animal health, increased financial viability and improved quality of life[6,11,12]. Rhode Island, Massachusetts, New Hampshire, Connecticut and Vermont are the top five states in the USA for direct marketing share of total agricultural sales and Maine ranks number seven. Three percent of total agricultural sales in New England are marketed directly to consumers[13].

New England also has a significant land base suitable for livestock production, with approximately 820,300 acres of pastureland and an additional 1,450,400 acres of cropland, some of which are used to grow animal feed crops[14]. Research by Peters et al. in New York State (which is nearly identical to New England in its ratio of cropland to pastureland[14]) showed that capacity to meet dietary requirements with local production increased with inclusion of livestock products in the diet, as these products utilize land limited to pasture and perennial forages[15].

However, small-scale livestock producers across the country have expressed difficulty in accessing slaughter and processing services[9]. A recent survey of livestock producers in southern New England (Connecticut, Rhode Island and Massachusetts) found that 70% of respondents would expand their business if they had better access to slaughter facilities[16]. A 2008 survey of Massachusetts livestock producers found that respondents were traveling an average of 52 miles each way to get to a slaughterhouse, for a total of 208 miles traveled when considering delivery and pick-up. Fifty-six percent of respondents said location was the most important quality in a slaughter facility, and 40% said that schedule availability during the fall season was most important. The majority of respondents sent very few (one to ten) animals to slaughter each year[17].

The decline in the number of small-scale slaughter facilities reflects the changing structure of the US livestock industry over the past 25 years[18]. Between 1980 and 2000, the industry became geographically concentrated in low-cost regions, with beef packing mostly in the Great Plains and pork processing primarily in the Southeast and Great Plains[19,20]. During this period, the number of slaughter plants nationwide dropped from over 600 to 170 for cattle and from over 500 to 180 for hogs[21]. Economists conclude that small-scale slaughter facilities have been squeezed out of the market due to new mechanical, biological and chemical technologies that enable larger firms to achieve significant economies of scale[18,21,22]. A USDA study found that costs were approximately 30% lower in 1,000,000-head-per-year plants than in 300,000-head-per-year plants[23]. These numbers dwarf the size of the industry in New England, where all 21 federally inspected plants combined slaughtered fewer than 15,000 cattle and 17,000 hogs in 2009[24].

Government food safety policy may also give larger firms a competitive advantage. The Hazard Analysis and Critical Control Points (HACCP) rule, promulgated in 1996, requires each plant to establish and implement its own sanitation standard operating plan (SSOP), conduct microbial testing, meet salmonella performance standards and identify critical control points in a detailed HACCP plan. The Government Accountability Office recognizes that the costs of HACCP plan implementation have disproportional effects on small plants, particularly diversified operations offering a range of processing services[25]. The slaughter industry in the six New England States is characterized by small-scale, family operated, diversified slaughter and processing facilities, the category of plant that has found it most difficult to keep up with increased costs[26].

The concerns of small-scale livestock producers in accessing slaughter and processing services have been well publicized in recent media activity, and many stakeholders seem eager for government intervention to revitalize local agricultural infrastructure[27,28]. However, there has been very little peer-reviewed research on New England's slaughter industry that could inform decisions on how best to expand or improve processing infrastructure. There has been some state-level research conducted in Vermont as part of the Farm-to-Plate strategic plan, which found that the state actually has underutilized slaughter (i.e. kill floor) capacity over the course of the year, and that the more significant pinch points are in cooling storage and processing (i.e., cut and wrap). A 2010 survey of Vermont slaughter facilities found that they operate at only 30–80% capacity during the off-season (February–August). The strategic plan concluded that, 'much can be done to address inefficiencies in the current slaughter and meat processing system, without having to significantly increase the total number of inspected meat-processing facilities in the state'[29].

Given the complexity of the small-scale livestock industry, more data are needed to understand the extent and causes of the presumed shortage of slaughter and processing capacity across the six New England states (Connecticut, Maine, Massachusetts, New Hampshire, Rhode Island and Vermont). In this study, we test the hypothesis that a shortage of large animal slaughter and processing infrastructure is a constraint to increased production of New England-produced meat by answering the following two questions:

(1) At what percent of maximum capacity are New England's large animal slaughter facilities operating?
(2) What percent of the total number of livestock produced in New England could be slaughtered in existing New England facilities?

We exclude poultry from this analysis because of the significant differences in regulatory regimes between the large animal and poultry slaughter industries. Furthermore, since exemptions exist for small-scale poultry producers that allow them to process their birds on farm, infrastructure constraints may have a more significant impact on the market availability of red meat (beef, pork, lamb, goat and veal) than poultry.

**Table 1.** Inspected large animal slaughter facilities in New England, May 2010[1].

| | Federally inspected | State-inspected | Total |
|---|---|---|---|
| Connecticut[2] | 3 | N/A | 3 |
| Maine | 6 | 7 | 13 |
| Massachusetts | 2 | N/A | 2 |
| New Hampshire | 1 | N/A | 1 |
| Rhode Island | 1 | N/A | 1 |
| Vermont | 7 | 1 | 8 |
| Total | 20 | 8 | 28 |

N/A = not applicable (states without state inspection programs).
[1] FSIS[30].
[2] All three of Connecticut's facilities have a federal inspection for slaughter only. Processing for these facilities is 'custom exempt', meaning that meat processed is not for resale.

## Methods

The analysis examined the total capacity of New England slaughter facilities, the utilization of available capacity and current livestock production. Each is addressed below.

The region's large animal slaughter facility owners and managers were surveyed to determine current capacity and identify challenges facilities face in meeting producer demand. A telephone questionnaire was designed and administered in March and April 2010. Survey participants were identified using a database of contact information for all USDA- and state-inspected slaughter facilities obtained from the Food Safety and Inspection Service (FSIS)[30]. A total of 28 facilities were identified in New England, 20 of which are federally inspected and eight of which are state-inspected (Table 1). Maine and Vermont have state-run inspection programs whose requirements have been deemed 'at least equal to' those imposed by FSIS.

Survey questions were grouped into three areas: (i) general information about the facility (questions about location, inspection status and number of employees); (ii) current physical capacity for slaughter and processing and current usage patterns; and (iii) current challenges and constraints to meeting farmer demand. The overall response rate was 82%. Survey information was used to estimate the total capacity for slaughter and processing for all New England plants over the course of a year. It was also used to glean a qualitative understanding of the industry.

Usage of New England's slaughter facilities was estimated based on data from the USDA National Agricultural Statistics Service *Livestock Slaughter Summary*[24] and the state inspection programs in Maine and Vermont (personal communication with Henrietta Beaufait, State Veterinarian, Maine Department of Agriculture, and Katherine McNamara, Assistant State Veterinarian, Vermont Agency of Agriculture, March 2010). Numbers of cattle, calves, hogs, sheep and goats slaughtered in all federal and state-inspected facilities were tabulated for New England. The total volume of livestock slaughtered was then compared to the estimates of New England's total slaughter capacity derived from the phone survey. Comparable data on livestock processing are not available from USDA, and thus, the assessment of capacity utilization for processing assumes that all animals slaughtered are also processed within the region. The analysis was completed using 2009 data for all livestock classes except for calves because of data anomaly. Specifically, the livestock slaughter data include values from a large calf-only facility, which opened and closed in the 2009 calendar year. Data from 2008 were used in this case.

**Table 2.** Relative rates[1] of slaughter and processing for large animals.

| | Slaughter | Processing |
|---|---|---|
| Cattle | 1 | 1 |
| Hogs | 3.77 | 5.22 |
| Sheep/goats | 4.18 | 5.87 |
| Calves | 2.56 | 3.32 |

[1] Cattle are used as the baseline. Values shown are the equivalent numbers of livestock which can be slaughtered or processed in the same period of time needed for one head of cattle.

Calculating regional capacity to produce livestock is challenging. Unlike crops, livestock production is not clearly tied to a discrete land base. Rather, animals may be raised on feed grown outside the region. Moreover, herds of livestock can expand (or contract) over time, making it difficult to attribute production to a single year. As a result, USDA agricultural statistics do not measure 'production' of livestock, but rather tabulate livestock 'marketings', animals sold for the slaughter market and younger animals shipped across state lines for feeding and breeding purposes. Our estimates of New England's capacity for livestock 'production' are based largely on these 'marketings' data. Data on meat animal marketings for cattle, calves, hogs and sheep were obtained from the 2009 NASS *Meat Animal Production Summary*[31]. Comparable data were not available for goats. Therefore, goat marketings were estimated based on inventory data from the *2007 Census of Agriculture*[9]. Livestock production data were compared with the capacity estimates to determine ability to meet a potential increase in regional demand for slaughter.

## Results

### *Analysis of New England's livestock slaughter and processing capacity*

Survey respondents were asked to estimate the number of animals that could be slaughtered and processed per day in their facilities. Responses were different for each livestock class. Since a mix of livestock types are slaughtered and processed in New England, the daily capacity estimates were used to normalize the number of animals slaughtered from head of livestock to 'animal units'. This allows a standardized estimate for a daily mix of livestock types. The average ratio of the capacity for cattle to the capacity

**Table 3.** Daily slaughter and processing capacity of survey respondents in animal units[1], by inspection type, 2009[2].

| Type of facility | Operation | Mean | Median | Range | Standard deviation | Standard error |
|---|---|---|---|---|---|---|
| Federal facilities[3] | Slaughter | 15.9 | 12.5 | 5 to 50 | 11.9 | 3.2 |
| | Processing | 7.0 | 6 | 1 to 20 | 4.3 | 1.1 |
| State facilities | Slaughter | 9.9 | 10 | 2 to 15 | 4.4 | 1.6 |
| | Processing | 4.6 | 5 | 2 to 8 | 2.2 | 0.8 |

[1] Cattle used as a proxy for animal units, as all species are normalized to cattle equivalence.
[2] To estimate 2009 slaughter capacity, each plant's self-reported daily capacity was multiplied by 250, the number of business days per year. Capacity estimates for 2009 do not include one federally inspected facility that opened in April 2010, or a facility that was open for only a portion of 2009, and was dedicated solely to calves.
[3] The five federal plants that declined to participate in the survey were assigned a capacity equal to the average capacity of the 15 plants for which data were collected.

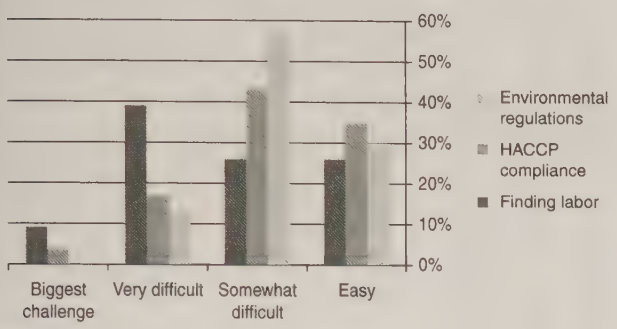


**Figure 1.** Attitudes towards potential constraints, percent of respondents.

for each other type of livestock for each plant was calculated to estimate equivalence. An analogous estimate was then made for processing. Values for the number of head per each animal unit are reported in Table 2. Because the relative capacity requirement for each livestock type is not consistent between slaughter and processing, animal unit equivalence is different for slaughter than it is for processing. The result is two different animal unit estimates, one for slaughter and one for processing.

Respondents' slaughter and processing capacity in animal units is reported in Table 3. At maximum physical capacity, federally inspected facilities could slaughter an average of 15.9 animal units per day and process an average of 7 units per day. State-inspected facilities had significantly less capacity, averaging a maximum physical capacity of 9.9 animal units for slaughter and 4.6 animal units for processing.

Seasonal demand fluctuations also varied among facilities, although all respondents reported being the busiest from late summer to early winter. Thirty-two percent said that reservations must be made 6–12 months in advance in order to ensure a slot between October and December, and 23% said that 6–12 months were needed for service between July and September. However, 41% of facilities could schedule customers within a month or less at any time during the year. Eighty-two percent could take customers within a month or less between April and June, and 86% could do so between January and March.

Participants were asked about the difficulty of finding labor, complying with environmental regulations and complying with HACCP regulations (Fig. 1). Labor was the most challenging of the three, with 48% of respondents saying it was very difficult to find labor, or that finding labor was their biggest challenge, in comparison with 19% for HACCP and 14% for environmental regulations. Thirty-five percent of respondents said that HACCP regulations were easy to comply with.

Only 18% said that they had more demand than they could meet at all times of the year. Nine percent said that they had insufficient demand during most times of the year. The majority (68%) reported adequate demand during most times of the year. When asked to identify their most significant challenge (as an open-ended question), the most common response (30% of respondents) was the seasonality of the industry, or maintaining business year round. Other common responses included the cost of insurance (13% of respondents) and obtaining credit (13% of respondents). A shortage of rendering companies, the cost of electricity, paperwork and dealing with inspectors were also identified as challenges (one respondent each).

## Comparing existing capacity to current utilization and production

The number of animal units slaughtered by type of livestock in both federally and state-inspected plants is reported in Table 4. The number of sheep and lambs slaughtered was nearly three times the number of cattle, which includes both beef animals and dairy culls. There were also a significant number of goats slaughtered in New England. State inspected facilities account for a small proportion of total slaughter, particularly for goats and calves.

Data for all types of livestock except for goats were reported monthly, and we see significant seasonal differences, with approximately 1700 animal units slaughtered in January, and over 3000 animal units slaughtered in October (Fig. 2). There is a spike in the number of sheep and lambs slaughtered in April, likely due to spring lamb sales for Easter (which fell on April 12 in 2009).

Meat production is reported in Table 5 in 2009 numbers of animals. Cattle and calves constitute the majority of livestock production in New England, with smaller production of hogs, sheep and goats. USDA does not track the

**Table 4.** Head slaughtered at New England federal[1]- and state[2]-inspected facilities, by class of livestock, 2009.

| Type of facility | Cattle | Hogs and pigs | Sheep and lambs | Goats | Calves[3] |
|---|---|---|---|---|---|
| | | | *No. of animals* | | |
| Federal-inspected | 13,300 | 16,800 | 36,200 | 28,869 | 2683 |
| State-inspected | 1555 | 1547 | 993 | 33 | 83 |
| Total | 14,855 | 18,347 | 37,193 | 28,902 | 2766 |

[1] Source of Federal statistics: NASS (2010)[24].
[2] Source of State statistics: VT Agency of Agriculture, Food and Markets and Maine Department of Agriculture, Food and Rural Resources.
[3] Federal data for calves are from 2008. Data for 2009 do not represent the historic trend, due to a calf slaughterhouse that was open for just a portion of the year.

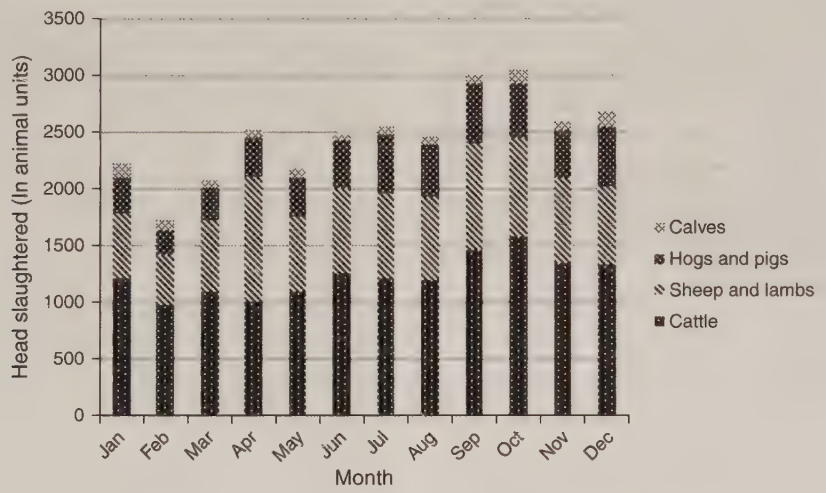


**Figure 2.** Head slaughtered by month, 2009, by class of livestock, in animal units.

number of calves sent directly to slaughter, as opposed to those sold as replacement heifers to dairy operations or feeder calves for beef production. To reflect this uncertainty in veal production, we report a high- and low-production estimate. The low estimate is the number of calves slaughtered in 2008, based on the assumption that the local veal market is already saturated, and production is not expected to increase significantly. The high estimate assumes that all beef calves are sold to feeding operations and that all dairy calves are sold for veal.

Slaughter and processing volumes are presented in Table 6, with comparisons to production by livestock category. Two estimates were calculated for processing capacity to account for three facilities that are inspected for slaughter only. A high estimate includes the three plants, demonstrating slaughter facilities' total on-site processing capacity for processing, were they to become inspected for processing in the future. A low estimate excludes the plants, demonstrating the region's inspected processing capacity only. The total capacity in animal units (equivalent to one head of cattle) is 95,071 for slaughter, 46,083 for on-site inspected and custom processing, and 39,000 for on-site inspected processing only.

When we convert the total number of head slaughtered in 2009 to animal units, we find that only 36,599 animal units were slaughtered in 2009. Comparing total slaughter capacity with number of animal units slaughtered in the region, we find that only 38.5% of New England's physical slaughter capacity was utilized in 2009.

Assuming that all of these animals were also processed within the region, the equivalent of 30,442 animal units was processed in 2009. Data are only collected for the number of head slaughtered, not processed, and so processing capacity utilization rates are estimates only. If we assume that all of the animal units slaughtered in the region were processed under inspection at slaughter facilities, 78.1% of facilities' processing capacity would have been utilized in 2009. However, this is likely a very high estimate, as it does not take into account animals that were processed at the three facilities without inspected processing or animals that were processed at facilities other than slaughterhouses (e.g., specialized cut-and-wrap operations, butcher shops or restaurants). If all on-site processing infrastructure is included in the capacity estimate (including custom processing), facilities operated at 66.1% of the total physical processing capacity.

Given the uncertainties in capacity and production noted above, Table 6 reports several estimates for the proportion of total livestock production that could be slaughtered and processed within the region. Assuming current levels of veal production (low production estimate), existing infrastructure could slaughter 84.6% of total production, and process 42.6% (assuming total physical on-site capacity) or 36.1% (assuming inspected on-site capacity only). Were all

**Table 5.** Estimated total meat production by class of livestock (head sold), 2009[1].

| State | Cattle | Hogs and pigs | Sheep and lambs | Goats[2] | Calves[3] | |
|---|---|---|---|---|---|---|
| | | | | | Low | High |
| Connecticut | 13,900 | 4100 | NA | NA | NA | 6904 |
| Maine | 15,700 | 9400 | NA | NA | NA | 10,347 |
| Massachusetts | 8000 | 10,200 | NA | NA | NA | 6351 |
| New Hampshire | 7800 | 6200 | NA | NA | NA | 4922 |
| Rhode Island | 1100 | 3400 | NA | NA | NA | 509 |
| Vermont | 47,000 | 5300 | NA | NA | NA | 66,257 |
| New England totals | 93,500 | 38,600 | 28,000 | 9364 | 2766 | 95,289 |

NA, no data available.
[1] Unless otherwise noted, production is the total number of animals marketed in 2009; Source: NASS (2010)[31].
[2] USDA-NASS does not include goat marketings in its annual survey data at the state or regional level. However, both sales and inventory data were collected in the 2007 Census of Agriculture. We assumed that the ratio of sales to inventory was consistent between 2007 and 2009, and used that ratio to estimate goat sales from 2009 inventory data.
[3] USDA-NASS estimates of the number of calves marketed include both beef and dairy calves. Considerable uncertainty exists around the fate of these marketed calves. The low estimate assumes that most are sold to dairy or cattle feeding operations outside New England. The high estimate assumes that all beef calves are sold to cattle feeding operations outside New England, whereas all dairy calves are sold for slaughter as veal. The number of dairy calves marketed for slaughter was estimated by multiplying the total number of calves marketed by the ratio of milk cows (that had calved) to total cows (that had calved).

**Table 6.** Slaughter and processing volumes and comparison with available infrastructure, by livestock category.

| Livestock category | Volume | | Comparison with available infrastructure | | |
|---|---|---|---|---|---|
| | (Animal slaughter units) | (Animal processing units) | (Slaughter capacity[1]) | (Inspected processing capacity[2]) | (Inspected and custom processing capacity[3]) |
| | | | Capacity utilization (%) | | |
| Animals slaughtered | 36,599 | 30,442 | 38.5 | 78.1 | 66.1 |
| | | | % of production that could be processed | | |
| Animals produced (low)[4] | 113,768 | 108,095 | 84.6 | 36.1 | 42.6 |
| Animals produced (high) | 149,875 | 135,927 | 63.4 | 28.7 | 33.9 |

[1] Slaughter capacity is estimated at 95,071 animal units. For this portion of the analysis, the newly opened plant was included for both slaughter and processing capacity.
[2] On-site, inspected processing capacity is estimated at 39,000 animal units. This estimate excludes the three facilities with only custom processing.
[3] Total on-site processing capacity (inspected and custom) is estimated at 46,083 animal units.
[4] Considerable uncertainty exists around the fate of marketed calves. The low estimate assumes that most are sold to dairy or cattle feeding operations outside New England. The high estimate assumes that all beef calves are sold to cattle feeding operations, whereas all dairy calves are sold for slaughter as veal.

dairy calves to be slaughtered within the region (high production estimate), existing infrastructure could slaughter 63.4% and process 33.9% (physical capacity) or 28.7% (inspected capacity).

## Discussion

Were the demand for locally produced red meat increased to such a degree that all of New England's livestock producers wanted their animals slaughtered and processed within the region, we find that current levels of production would outstrip existing slaughter facilities' slaughter and processing capacity. However, the degree of the capacity shortage varies with different scenarios for processing capacity and calf production, and assumes significant increases in local marketing for some species. For instance, there are many more cattle, calves and hogs marketed than slaughtered in New England. No data are available on where the end markets are, but for cattle and hogs the majority are likely sold at auction in New York and Pennsylvania and slaughtered for the commodity market (telephone interview with J. Smith, University of Vermont Extension Dairy Specialist, July 7, 2010). Many of these producers are not interested adjusting their business models to market their meat locally. It is also interesting to note that for sheep the number slaughtered is actually higher than the number marketed, indicating that animals from outside of the region are being slaughtered in New England.

Perhaps surprisingly, we find that currently the region's slaughter capacity is actually substantially underutilized over the course of a year. One key reason for the low-capacity utilization rate seems to be the seasonality

of the industry, with the highest levels of demand for slaughter and processing occurring in the late summer and fall. The demand for slaughter and processing are low during the winter and spring months for several reasons. First, spring is the traditional calving season for beef cattle in New England, and the average time from calf to market weight is 18 months, such that the animals are ready for slaughter in the fall.

There are also incentives to keep a small herd in the winter. For ruminants in grass-based production systems, the price of feeding increases significantly over the winter, when farmers must either buy feed or store what they have grown during the season (and absorb associated storage losses). For all livestock, feed efficiency decreases in the colder months as animals spend more calories on body temperature maintenance[32]. Finally, producers that engage in direct marketing often have more opportunities to sell their meat in the summer, when farmers, markets and farm stands are open. For these reasons, slaughter facilities often have less business in the spring. When asked to report their biggest challenge in an open-ended question, one-third of facilities discussed seasonality and a lack of year-round demand.

A shortage of labor compounds seasonal constraints on infrastructure availability by limiting facilities' capacity to temporarily ramp up production during the busy season. Unlike farms, which can hire unskilled labor on a seasonal basis, large animal slaughter and processing facilities require a more skilled labor force. There are also fewer people willing to do the work. Because of the resources needed to develop human capital, and the small size of the labor pool, slaughter facilities avoid laying employees off during the slow season. One survey respondent reported that he was paying his crew to paint his house because he did not want to recruit and train new employees in the summer.

Plant location could be another important reason for underutilization, and spatial analysis could determine areas where producers are isolated from slaughter and processing infrastructure. More research is also needed on the appropriate transportation distance from farm to slaughter. Producers on small, diversified livestock operations may be more transportation-adverse than those on large, specialized operations due to higher opportunity costs of time spent on the road, and an inability to achieve economies of scale for transport. Existing research on transportation's economic costs, carbon costs, and effects on animal health should be analyzed to estimate maximum feasible distance. These factors must be balanced against slaughter facilities' need to maintain sufficient throughput to achieve financial viability, given their slim profit margins.

Furthermore, there is a significant difference between the slaughterhouses' capacity for slaughter versus processing. Traditionally, processing was not under the purview of the slaughterhouse, especially for beef. In the 1970s most facilities sold whole or half carcasses to retailers who would then cut them down into pieces, often to order for each customer[23]. With the rise of supermarkets and convenience foods, retailers preferred to receive cuts of meat, which could be more easily stored and marketed than an entire side of beef.

More research is also needed to determine the total capacity for processing in the region, including processing being done outside of slaughter facilities. Examining processing separately from slaughter could be an effective way to increase capacity. The inspection requirements are not as burdensome for processing as for slaughter, and the potential labor pool is larger. There is evidence of a renaissance of artisanal butchery, with a growing number of chefs buying whole animals and cutting them down in the restaurant to economically meet the demand for local meat. A 2009 New York Times article went so far as to deem butchers the rock stars of the food world[33]. Were a greater proportion of processing to occur outside of slaughterhouses, better transportation systems would be required such that producers would not be required to make an additional trip from slaughter to processing facility. For instance, the slaughterhouse could ship the carcass to the processor, increasing the total cost of processing, but perhaps increasing access to a greater range of value-added processing services.

## Conclusion

A more complete understanding of the meat processing industry is needed in order to determine how to improve or expand capacity and promote growth in local red meat production. While we foresee a long-term need for additional infrastructure, any efforts to increase the number of slaughter facilities must address the competitive nature of the national meatpacking industry and a regulatory regime that may be disproportionally burdensome for small-scale plants. We find that the infrastructure exists in New England to slaughter most of the livestock produced in the region, but a relatively small percentage of this existing capacity is currently utilized. While further research may demonstrate that spatial separation between producers and slaughterhouses necessitates the construction of new plants closer to production loci, there may be ways to maximize utilization of current infrastructure in the short term. Increasing regional meat production requires more than simply increasing slaughter and processing infrastructure.

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# Soil fertility and crop yields in long-term organic and conventional cropping systems in Eastern Nebraska

Sam E. Wortman*, Tomie D. Galusha, Stephen C. Mason, and Charles A. Francis

Department of Agronomy and Horticulture, University of Nebraska-Lincoln, Plant Science Hall 279, Lincoln, NE 68583, USA.
*Corresponding author: sam.wortman@huskers.unl.edu

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## Abstract

Organic agriculture aims to build soil quality and provide long-term benefits to people and the environment; however, organic practices may reduce crop yields. This long-term study near Mead, NE was conducted to determine differences in soil fertility and crop yields among conventional and organic cropping systems between 1996 and 2007. The conventional system (CR) consisted of corn (*Zea mays* L.) or sorghum (*Sorghum bicolor* (L.) Moench)–soybean (*Glycine max* (L.) Merr.)–sorghum or corn–soybean, whereas the diversified conventional system (DIR) consisted of corn or sorghum–sorghum or corn–soybean–winter wheat (wheat, *Triticum aestivum* L.). The animal manure-based organic system (OAM) consisted of soybean–corn or sorghum–soybean–wheat, while the forage-based organic system (OFG) consisted of alfalfa (*Medicago sativa* L.)–alfalfa–corn or sorghum–wheat. Averaged across sampling years, soil organic matter content (OMC), P, pH, Ca, K, Mg and Zn in the top 15 cm of soil were greatest in the OAM system. However, by 2008 OMC was not different between the two organic systems despite almost two times greater carbon inputs in the OAM system. Corn, sorghum and soybean average annual yields were greatest in either of the two conventional systems (7.65, 6.36 and 2.60 Mg ha$^{-1}$, respectively), whereas wheat yields were greatest in the OAM system (3.07 Mg ha$^{-1}$). Relative to the mean of the conventional systems, corn yields were reduced by 13 and 33% in the OAM and OFG systems, respectively. Similarly, sorghum yields in the OAM and OFG systems were reduced by 16 and 27%, respectively. Soybean yields were 20% greater in the conventional systems compared with the OAM system. However, wheat yields were 10% greater in the OAM system compared with the conventional DIR system and 23% greater than yield in the OFG system. Alfalfa in the OFG system yielded an average of 7.41 Mg ha$^{-1}$ annually. Competitive yields of organic wheat and alfalfa along with the soil fertility benefits associated with animal manure and perennial forage suggest that aspects of the two organic systems be combined to maximize the productivity and sustainability of organic cropping systems.

**Key words:** long-term crop rotations, organic farming, animal manure, perennial forage, organic matter content, soil phosphorus, nutrient budgets

## Introduction

Spurred on by concerns about increasing farm size, environmental pollution, reduced biodiversity across the landscape and increased consumer demand for organic products, investigators have reported on long-term studies of differences between conventional and organic cropping systems[1–6], often with contradictory results. Compared with conventional cropping systems, reduced yields were reported for most crops in organic systems[1,2,5,6], whereas some organic systems produced equal or greater yields[3]. Yield differences between the two cropping systems are often crop specific. For example, wheat yields are often comparable in organic and conventional cropping systems[5,6]. During the first four cycles of a long-term crop rotation experiment (1975–1991) in east central Nebraska, there were no significant differences in corn or soybean yields between conventional and organic systems in a four-year rotation; moreover, corn yields were consistently greater in rotation than in continuous corn[7].

Complex, diverse and extended crop rotations are an essential component of sustainable crop production[8]. Organic cropping systems rely on diverse rotations to provide pest control and increase crop nutrition through biological

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nitrogen fixation and recycling of nutrients[9]. Additionally, organic cropping systems often rely on organic soil amendments such as farmyard manure or compost. Both crop rotation and manure amendments have been shown to improve grain yield compared with continuous cropping systems with synthetic fertilizers[10]. Moreover, organic soil amendments have been associated with improved soil properties such as increased soil organic matter content (OMC) and water-holding capacity, lower bulk density and enhanced pH stabilization[3,11–13]. Soil organic matter is thought to drive the productivity and sustainability of organic systems as it provides a nitrogen reservoir and increases soil water-holding capacity[3,14]. Animal manure amendments also have been shown to increase soil levels of essential nutrients including Ca, Mg, K and P[15,16].

Including perennial forage such as alfalfa in rotation may improve crop yields, soil structure, OMC and nutrient cycling[17]. Alfalfa is particularly desirable due to its capacity to biologically fix atmospheric N. In a recent study, corn grain yield in an extended rotation with alfalfa and no additional N input was not different from corn supplemented with synthetic N fertilizer; the leguminous forage provided sufficient N for the subsequent corn crop. However, soil P levels decreased in the rotation with alfalfa, presumably the result of crop removal[18]. Organic systems that include forage removal (e.g., alfalfa) and limited manure or compost application may reduce available P below critical crop response values, depending on initial soil P balance, yield of removed grain and forage, and frequency and rate of manure application[19]. Over time, grain-based annual cropping systems may become N-limited, while rotations that include perennial forage and grain crops may become P-limited[19]. Therefore, a combination of legumes, forages and organic amendments such as animal manure may be required for the maintenance of soil nutrients and OMC in organic cropping systems.

In 1975, the Long-Term Crop Rotation experiment was initiated at the Agricultural Research and Development Center (ARDC) near Mead, NE to determine the effects of crop rotation and animal manure on grain yield. The first four cycles of the rotation were reported by Lesoing[7] (stage one of the long-term experiment). Redesigned in 1996, the experiment now includes a forage-based organic cropping system in the comparisons between organic and conventional cropping systems (stage two of the long-term experiment). Stage two of the experiment was designed from a 'systems-based' perspective, with the goal of understanding some of the complex differences between realistic organic and conventional cropping systems. While this experimental approach offers practical insight regarding these different cropping systems, it does limit the discovery of mechanistic 'cause and effect' relationships typical of factorial experiments[20].

Objectives of this study were to determine the effects of long-term management and cropping system on soil chemical and physical properties and how these factors contribute to grain yield. We hypothesized that continued application of semi-composted bovine manure will: (1) maintain or increase the already greater levels of soil nutrients (P, Ca, Mg, K and Zn), soil pH and OMC established prior to 1996; and (2) increase grain yields when compared with an organic forage-based cropping system. Furthermore, we hypothesized that (3) the conventional cropping systems will have greater grain yields than either organic cropping system; (4) in years of less than average rainfall, yields will decrease less in cropping systems with high soil OMC; and (5) grain yields will increase with soil P and OMC.

## Materials and Methods

### *Cropping systems*

The Long-Term Crop Rotation experiment was initiated in 1975 at the University of Nebraska ARDC near Mead, NE, and redesigned in 1996 to evaluate the productivity of two organic and two conventional cropping systems that differ in crop diversity, pest management and nutrient input strategies. Dominant soil type at the site is a Sharpsburg silty clay loam (fine, smectitic, mesic Typic Argiudoll) with 0–5% slopes[7]. Total area of the study site is 4 ha and has been managed without irrigation.

The experiment was designed as a randomized complete block with four replicate blocks and 13 experimental units per block. From 1975 to 1995 (stage one) the treatments included continuous corn with synthetic fertilizer and herbicides (HFI CC), a 4-year rotation with synthetic fertilizer and herbicides (HF, corn–soybean–corn–oat/clover), the same four-year rotation with manure only (ORG), and the same 4-year rotation with only synthetic fertilizers (FO). The four cropping systems established within each block in 1996 (stage two) included: conventional (CR; to replace HFI CC), diversified conventional (DIR; to replace HF), organic animal manure-based (OAM; to replace ORG) and organic forage-based (OFG; to replace FO). While the management practices and cropping systems differ between the two stages of the experiment, soil analyses were conducted at the conclusion of stage one to document initial differences in soil properties attributed to management during stage one of the long-term experiment. The current DIR, OAM and OFG systems are replicated four times within each block so that each crop phase of the four-year crop sequence is present each year within each block. The CR system is present in only one experimental unit per block with a single-crop phase. Each experimental unit in the original study was 0.047 ha (12.2 m × 38.4 m), and was split into half in 1996 to create two sequences within each crop rotation to include sorghum in the cereal phase of each rotation. In 1996, sorghum was an important agronomic crop in Nebraska representing 1.25 million acres of annual production[21]. Moreover, it was hypothesized that including sorghum in the rotations would increase the economic resilience of the systems, and potentially reduce weed populations through allelopathic interactions.

**Table 1.** Rotational sequences for each cropping system in stage two of the Long-Term Crop Rotation experiment at the University of Nebraska–Lincoln Agricultural Research and Development Center near Mead, NE. Cropping systems include conventional (CR), diversified conventional (DIR), organic animal manure-based (OAM) and organic forage-based (OFG).

| System | Sequence | 1st year | 2nd year | 3rd year | 4th year |
|---|---|---|---|---|---|
| CR | 1 | Corn | Soybean | Sorghum | Soybean |
| | 2 | Sorghum | Soybean | Corn | Soybean |
| DIR | 1 | Corn | Sorghum | Soybean | Wheat |
| | 2 | Sorghum | Corn | Soybean | Wheat |
| OAM | 1 | Soybean | Corn | Soybean | Wheat |
| | 2 | Soybean | Sorghum | Soybean | Wheat |
| OFG | 1 | Alfalfa | Alfalfa | Corn | Wheat |
| | 2 | Alfalfa | Alfalfa | Sorghum | Wheat |

Detailed crop rotation and sequences for the four cropping systems are summarized in Table 1. Cropping systems and rotation sequences were designed based on the input of a farmer advisory panel, which also advised on common cultural practices among conventional and organic farmers in eastern Nebraska prior to the 1996 growing season. The 'systems-based' experimental approach provided flexibility in the design of realistic organic and conventional cropping systems. Each of these rotations was built around corn and sorghum, as grain crop production dominates the eastern Nebraska landscape[21]. These two crops are present in all four rotations to allow for a common comparison among all systems, while soybean and winter wheat are present in at least one organic (OAM or OFG) and one conventional (CR or DIR) system to allow for comparison between organic and conventional systems in general.

Alfalfa was included in the OFG system because it represents a valuable commodity for livestock production, and is also seen as a potentially low-input crop well suited to organic production. While this commodity is traditionally utilized on-farm, it is not uncommon to export high-quality alfalfa hay from the farm (e.g., dairy farms, feedlots and ranches). The grain crops in the OAM system rely exclusively on animal manure for nutrient inputs, which is not an unrealistic option for most farmers in eastern Nebraska as there is an abundance of cattle feedlot operations[21]. Of course, an integrated crop–livestock operation would have the added flexibility of feeding a forage crop on-farm and recycling those nutrients through a free on-farm manure source. However, as farming operations become less diverse and more specialized in crop or livestock production, the reality of exporting nutrients (in the form of feed or manure) from the farm is increasingly common[22].

## Corn and sorghum

In the CR and DIR systems, each spring corn and sorghum experimental unit was fertilized with an average of 120 kg N $ha^{-1}$ (as urea, ammonium nitrate, liquid N or anhydrous ammonia), and an average of 40 kg P $ha^{-1}$ (broadcast applied as dry superphosphate between 1997 and 1999). Annual fertilizer application rates were determined after consideration of soil tests from select experimental units (i.e., few experimental units sampled with insufficient sample size for statistical analysis), realistic crop yield goals and University of Nebraska recommendations[23]. In most years, seedbed preparation consisted of disking and/or field cultivation. Average corn seeding rate was 57,300 seeds $ha^{-1}$ and average sorghum seeding rate was 321,000 seeds $ha^{-1}$. Weed management for corn and sorghum in the conventional systems included a diverse selection of pre- (PRE) and post-emergence (POST) herbicides combined with two inter-row cultivations.

Corn and sorghum in the OAM system were amended with an average of 31.7 Mg $ha^{-1}$ (dry matter basis hereafter) of semi-composted bovine manure, while the OFG system received just one 31.3 Mg $ha^{-1}$ application of semi-composted bovine manure in 1996 to alleviate potential P deficiency (Table 2). Based on a 12-year average analysis of the bovine feedlot manure source utilized in the organic systems, total N content was 1.16%, total P content was 0.68% and total K content was 2.43% (Table 2). Manure was broadcast applied and incorporated with a field disk in the spring. Similar to the synthetic fertilizer rates, manure application rates for all crops were determined after consideration of soil tests from select experimental units within each system, and realistic crop yield goals. However, manure application rates in the OAM system were based on crop N demands, while rates in the OFG system were typically based on potential P deficiency in subsequent crops. Between 1997 and 1999, a hairy vetch (*Vicia villosa*) living mulch was included during the corn and sorghum phase of the OFG system (34 kg $ha^{-1}$ broadcast seeding rate; approximately 1.1 million seeds $ha^{-1}$). Hairy vetch seed was broadcast with hand-held spreaders during early vegetative growth of corn and sorghum. However, this practice was discontinued after 1999 due to poor establishment and growth of the hairy vetch. Both organic systems were disked and field cultivated each spring for seedbed preparation. The seeding rates for corn and sorghum in the organic systems were the same as those in the conventional systems. Weed management for corn and sorghum in the organic systems included an average of three early cultivations with a rotary hoe and two inter-row cultivations.

Corn was typically planted between late-April and mid-May in 0.76 m rows, while sorghum was planted mid- to late-May in 0.76 m rows. Corn hybrids planted include: Hoegemeyer Hybrids (Hooper, NE, USA) '2591', '2598', '2647', '2648', '2649', '2680' and '2693'; NC+ (Lincoln, NE, USA) '3448', '40M21', '54M52' and '69R36'; Pioneer (Johnston, IA, USA) '3751'; DeKalb (Thomasboro, IL, USA) '646'; and 33 different hybrids from Blue River Hybrids (Kelley, IA, USA). Between 2004 and 2007 CR and OAM experimental units were utilized in a genotype by system variety trial using Blue River Hybrids corn and soybean. These experimental units were divided into 15

**Table 2.** Year, cropping system, crop phase, rates and nutrient analysis (dry matter basis) for semi-composted bovine feedlot manure applications utilized in the OAM and OFG systems between 1996 and 2007 in the Long-Term Crop Rotation experiment near Mead, NE. Year is relative to a given crop phase, not the actual application (e.g., 2002 OFG-A had manure applied in the fall of 2001, but is intended for crop uptake during the 2002 crop phase). OAM, organic animal manure-based system; OFG, organic forage-based system; CS, corn/sorghum phase; WW, winter wheat phase; A, alfalfa phase.

| Year System-phase | N | P | K | Application rate |
|---|---|---|---|---|
| | --------%-------- | | | ----Mg ha$^{-1}$---- |
| 1996 | | | | |
| OAM–CS | 1.76 | 0.77 | 4.00 | 13.1 |
| OFG–CS | 1.76 | 0.77 | 4.00 | 32.4 |
| OAM–WW | 0.88 | 0.29 | 2.47 | 38.0 |
| OFG–A | 0.92 | 1.19 | 2.46 | 31.3 |
| 1997 | | | | |
| OAM–CS | 0.94 | 0.55 | 2.49 | 25.3 |
| OAM–WW | 1.27 | 0.66 | 2.29 | 20.8 |
| 1998 | | | | |
| OAM–CS | 0.68 | 0.82 | – | 58.5 |
| OAM–WW | 0.68 | 0.82 | – | 5.8 |
| 1999 | | | | |
| OAM–CS | 0.59 | 0.54 | 2.47 | 80.7 |
| OAM–WW | 5.66 | 0.35 | 2.37 | 44.8 |
| OFG–A | 0.85 | 0.79 | 3.41 | 60.6 |
| 2000 | | | | |
| OAM–CS | 0.41 | 0.50 | 2.61 | 19.7 |
| OAM–WW | 0.41 | 0.50 | 2.61 | 38.8 |
| OFG–A | 0.55 | 0.45 | 2.82 | 57.3 |
| 2001 | | | | |
| OAM–CS | 1.36 | 0.97 | 2.15 | 23.0 |
| OAM–WW | 1.36 | 0.97 | 2.15 | 20.0 |
| OFG–A | 0.41 | 0.50 | 2.61 | 35.4 |
| 2002 | | | | |
| OAM–CS | 1.74 | 0.46 | 2.81 | 44.6 |
| OAM–WW | 0.61 | 0.62 | 1.31 | 12.1 |
| OFG–A | 0.61 | 0.62 | 1.31 | 45.2 |
| 2003 | | | | |
| OAM–CS | 1.13 | 0.54 | 1.30 | 22.8 |
| OAM–WW | 1.13 | 0.54 | 1.30 | 22.8 |
| 2004 | | | | |
| OAM–CS | 1.13 | 1.22 | 1.57 | 25.9 |
| 2005 | | | | |
| OAM–CS | 1.46 | 0.87 | 2.61 | 14.9 |
| OFG–A | 1.46 | 0.87 | 2.61 | 20.2 |
| 2006 | | | | |
| OAM–CS | 0.98 | 0.60 | 2.53 | 25.7 |
| 2007 | | | | |
| OAM–CS | 0.60 | 0.47 | 2.51 | 25.9 |

sub-plots and grain yield for this study was reported as the average of all 15 sub-plots. Genotypes tested in this variety trial represented competitive and modern hybrids/varieties for eastern Nebraska; thus, the range of yield differences among genotypes within systems was reasonably small (unpublished data). Compiling the average yield for these genotypes was seen as the most representative sample of yield for the entire experimental unit. Sorghum varieties planted include: DeKalb '28E', '39Y', '42-20', '53' and '53-11'; NC+ '6B67' and '7R37E'; and Pioneer '87G57'.

### Soybean

In the CR and DIR systems, corn and sorghum stalks were shredded each spring and experimental units were disked and/or field cultivated for seedbed preparation. Between 1997 and 2003, soybean in the CR system was planted with a drill in 0.25 m rows at an average rate of 680,000 seeds ha$^{-1}$, while in 1996 and between 2004 and 2007 soybean was planted in 0.76 m rows at an average rate of 470,000 seeds ha$^{-1}$. Soybean in the CR system was transitioned out of 0.25 m rows in 2004 to establish the Blue River Hybrids genotype × system variety trial. Soybean in the DIR system was always planted with a drill in 0.25 m rows at an average rate of 642,000 seeds ha$^{-1}$. Weed management for soybean in the conventional systems included a diverse selection of PRE and POST herbicides combined with one manual roguing operation and two inter-row cultivations when 0.76 m rows were used in the CR system. Glyphosate-resistant soybean varieties were always used in the DIR system and CR system (1997–2003) to allow for POST applications of glyphosate.

In the OAM system, corn and sorghum stalks were shredded each spring and experimental units were disked and field cultivated twice for seedbed preparation. Soybean in the OAM system was planted in 0.76 m rows at an average rate of 470,000 seeds ha$^{-1}$. Weed management for soybean in the OAM system included an average of three early cultivations with a rotary hoe, two inter-row cultivations and two manual roguing operations.

Timing of soybean planting was typically mid to late May for all systems. Soybean varieties planted include: DeKalb '28-32', and '28-52RR'; Hoegemeyer Hybrids '202', '271RR' and '274'; NC+ '2F11'; Pioneer '9233', '9306', '92B36RR', '92B38RR', '92B61' and '92B63'; UNL Foundation Seed (Ithaca, NE, USA) '3100', 'Conrad', 'Dunbar', 'Hamilton', 'Holt' and 'Resnik'; and 40 varieties from Blue River Hybrids (genotype by system variety trial). Similar to corn experimental units between 2004 and 2007, the CR and OAM soybean experimental units were divided into 15 sub-plots and grain yield for this study was reported as the average of all 15 sub-plots.

### Wheat

The DIR system was fertilized with an average of 70 kg N ha$^{-1}$ (as ammonium nitrate or liquid N) and 60 kg P ha$^{-1}$ (as dry superphosphate). Wheat was planted with a drill in 0.25 m rows at an average rate of 134 kg ha$^{-1}$ (approximately 4.5 million seeds ha$^{-1}$). Weed management for wheat in the DIR system was limited to PRE herbicide applications in 2003 and 2005. Following wheat harvest, wheat stubble and weeds in the DIR system were managed

with glyphosate application, mowing, disking and/or plowing.

In the OFG system, corn and sorghum stalks were shredded and disked twice for seedbed preparation prior to planting wheat. The OAM system was disked or field cultivated once for seedbed preparation. Each spring between 1996 and 2003 the OAM system was amended with an average of 25.4 Mg ha$^{-1}$ of semi-composted bovine manure (Table 2). This manure was applied over-the-top of the wheat crop and was not incorporated. Wheat in the organic systems was also planted with a drill in 0.25 m rows at an average rate of 134 kg ha$^{-1}$ (approximately 4.5 million seeds ha$^{-1}$). Following harvest, wheat stubble and weeds in the organic systems were managed with mowing, disking, field cultivation and/or plowing. In six of 12 study years, wheat stubble in the OFG system was amended with an average of 41.7 Mg ha$^{-1}$ of semi-composted bovine manure to alleviate potential P deficiency in the subsequent alfalfa crop (Table 2). These manure applications were incorporated with a disk during simultaneous weed control operations.

For the DIR, OFG and OAM systems, wheat was typically planted between mid October and mid November. Wheat varieties planted include: UNL Foundation Seed '2137', 'Karl', 'Millenium', 'Nekota', 'Wahoo', plus eight additional varieties from UNL Foundation Seed. In 2002 and 2003, all wheat experimental units were utilized in a genotype by system variety trial. Similar to the corn and soybean variety trials, experimental units were divided into eight sub-plots and grain yield for this study was reported as the average of all eight sub-plots in 2002 and 2003.

### *Alfalfa*

Alfalfa was planted at 28 kg ha$^{-1}$ (approximately 12.3 million seeds ha$^{-1}$) in 0.25 m rows, typically between early August and late September. Alfalfa was cut and baled three times per season and incorporated with a moldboard plow at the end of the second season. Alfalfa varieties planted include: Crow's (Kentland, IN, USA) 'Synergy'; NC+ 'Jade'; and Blue River Hybrids 'Bluebird', 'Jade II' and 'Meadowlark'.

### *Data collection*

There were no records of soil test analyses at the initiation of the experiment in 1975, but it is assumed that the soil nutrient levels of the field were relatively uniform as the field was in continuous alfalfa for three years prior to 1975. Since the beginning of the current cropping systems in 1996, soil samples were taken at the beginning of each 4-year rotation cycle, in early spring of 1996, 2000, 2004 and 2008. Starting in 1996, corn plots were split into half to accommodate equal-sized subplots of corn and sorghum; soil samples were always taken from the corn half of each plot at the beginning of a given cycle. Soil samples were collected to a depth of 122 cm with a tractor-mounted Giddings hydraulic soil coring and sampling machine (Giddings Machine Co., Windsor, CO, USA). Cores were divided into four depths (0–15, 15–61, 61–91 and 91–122 cm), and only the surface depth (0–15 cm) was analyzed in this study. Two samples were taken in random locations within each split-plot experimental unit and pooled to form one composite sample. The sample was mixed by hand and was air dried. A subsample was then boxed without sieving and was delivered to Ward Laboratories Inc. (Kearney, NE, USA) for analysis of soil pH, OMC, P, K, Ca, Mg, Na, $SO_4$ and Zn. Soil bulk density data were unavailable for each experimental unit; thus, results from the OMC analysis should be interpreted with caution as the integrity of these values depends on the assumption of equal bulk densities within and among treatments.

Soil extraction and analyses were conducted according to routine laboratory procedures at Ward Laboratories Inc. Soil pH was determined using a 1 : 1 soil–water ratio, and OMC was measured as percent oxidizable organic carbon in soil using the Walkley–Black method[24,25]. Phosphorus was extracted by the Bray-1 P test, and the extracting solution consisted of 0.025 M HCl and 0.03 M $NH_4F$[26]. Potassium (K), Ca, Na and Mg cations were each extracted with 1 N ammonium acetate. The soil and extracting solution were mixed vigorously for 5 min, and the soil was then filtered from the liquid. Individual cation concentrations in the filtrate were analyzed by flame emission in an atomic absorption spectrophotometer[24]. Sulfate was extracted using a calcium phosphate (500 ppm P) extractant. The soil extractant was mixed vigorously for 30 min, and $SO_4$ concentration in the filtrate was determined by developing a barium sulfate turbidity (determined by flow injection analysis)[24]. Zn was extracted (2 h) with a diethylenetriaminepentaacetic acid (DTPA) chelate solution, which included 0.005 M DTPA, 0.01 M $CaCl_2$ and 0.1 M triethanolamine (TEA) (pH = 7.3)[24].

Growing season precipitation and temperature for corn, sorghum and soybean (April 1–September 30) and winter wheat (October 1–June 15) were determined using the sum of daily precipitation and temperature measurements from the High Plains Regional Climate Center station located on the University of Nebraska Turf Farm near Mead, NE (41°10′12″N lat., 96°28′12″W long., elevation = 366 m), located 1 km northwest of the rotation experiment (Table 3). Climate data for the 30-year mean were obtained from a different climate center near Mead, NE (41°8′24″N and 96°28′48″W) between 1971 and 2000 (long-term data from the University of Nebraska Turf farm were unavailable).

Grain crop yield was measured by harvesting the middle 4.6 m × 38.4 m in each experimental unit, transferring the combine contents into a weigh wagon and measuring grain moisture content in the laboratory with a GAC 2100 grain moisture tester (DICKEY-john Grain Instrumentation, Ankeny, IA, USA). Corn grain yields were adjusted to 0.155, sorghum to 0.140 and soybean and wheat to 0.130 g kg$^{-1}$ moisture. Alfalfa yield was measured by collecting three biomass samples from three separate

**Table 3.** Annual growing season precipitation and temperature values for summer annual (April 1–September 30) and winter annual (October 1–June 15) crops between 1996 and 2007 measured 1 km northwest of the Long-Term Crop Rotation Experiment near Mead, NE. The 30-year mean represents the mean precipitation and temperature near Mead, NE (41°8′24″N and 96°28′48″W) between 1971 and 2000.

| | Precipitation (mm) | | Temperature (°C) | |
|---|---|---|---|---|
| Year | April 1–September 30 | October 1–June 15 | April 1–September 30 | October 1–June 15 |
| 1996 | 499 | 355 | 18.0 | 4.7 |
| 1997 | 378 | 189 | 18.1 | 5.7 |
| 1998 | 530 | 566 | 20.1 | 7.3 |
| 1999 | 600 | 463 | 18.8 | 7.2 |
| 2000 | 401 | 226 | 19.6 | 6.5 |
| 2001 | 452 | 531 | 19.6 | 6.9 |
| 2002 | 470 | 337 | 19.8 | 6.4 |
| 2003 | 363 | 367 | 18.6 | 6.0 |
| 2004 | 451 | 444 | 18.6 | 6.6 |
| 2005 | 397 | 424 | 19.9 | 6.8 |
| 2006 | 506 | 335 | 19.6 | 7.7 |
| 2007 | 610 | 524 | 19.6 | 6.0 |
| Study mean | 471 | 397 | 19.2 | 6.5 |
| 30-year mean | 519 | 411 | 19.0 | 5.8 |

windrows (harvest area of 1.52 m × 4.88 m), drying at 60°C to constant mass, and weighing. In 1997, wheat was harvested for hay (due to severe drought and broadleaf weed pressure) and yield was determined using the alfalfa yield determination method.

### *Statistical analysis*

Every 4 years of a rotation was considered a full cycle and within a given cycle there were four crop phases. For example, cycle one included crops grown between 1996 and 1999, while crop phase one included crops grown in 1996, 2000 and 2004 in a given experimental unit and crop phase two included crops grown in 1997, 2001 and 2005, etc. Therefore, between 1996 and 2007 there were three complete rotation cycles and four crop phases within each cycle. This analysis allows comparison of each crop at the same point in each rotation sequence. This experimental design is a split-plot over time, with cropping system (e.g., OFG) as the whole plot and year/current crop (e.g., 1999 corn) as the split-plot[7]. Soil sampling always occurred in the corn half (sequence 1) of all experimental units during crop phase one of each rotation cycle; thus, statistical analysis involving soil properties did not include crop phase or sequence as fixed effects.

While the CR system was only present in one whole plot experimental unit per block, comparisons with the three other systems were still possible within each crop phase for corn, sorghum and soybean. For example, in 1996 corn and sorghum split-plots were present in CR blocks 1 and 3 (crop phases 1 and 3 are present in both blocks due to split-plot experimental units), while soybean was present in CR blocks 2 and 4 (crop phases 2 and 4). Crop phases 2 and 4 were eventually pooled for analysis due to the lack of differences following corn (crop phase 1) and sorghum (crop phase 3). Thus, yield means in the CR system were available for comparison in all years, but the variance was inflated due to the reduced number of replications (two for corn, sorghum and soybean phases) in this system.

Analysis of variance was conducted to determine differences in individual crop yield. Fixed effects in the model included crop phase, system, year(crop phase) (crop phase 'nested' within year due to the dependence of crop phase on year), sequence, and the two-way interactions of crop phase × system, and year(crop phase) × system. Random effects included block (crop phase) and the interactions of system × block(crop phase) and year × block(crop phase) (PROC MIXED; SAS Version 9.1, SAS Inst., Cary, NC, USA). Yields were then pooled by crop phase, sequence and year within systems and compared using orthogonal contrasts. Similarly, analysis of variance was used to determine the effect of growing season precipitation on crop yield; this model was identical to the model outlined above except that growing season precipitation replaced year as a fixed effect and crop phase was not nested within the effect of precipitation. Crop phase and sequence were again pooled within systems and compared using orthogonal contrasts.

Analysis of variance was also used to assess the effects of system, OMC and P on crop yield. Fixed effects in this model included system, OMC, P and the two-way interactions of system × OMC and system × P. Block and the block × system interaction were the random effects in the model. Lastly, differences in individual soil properties were determined with analysis of variance where system, previous crop and year were considered fixed effects and block and the block × system interaction were the random effects in the model. Previous crops were then pooled

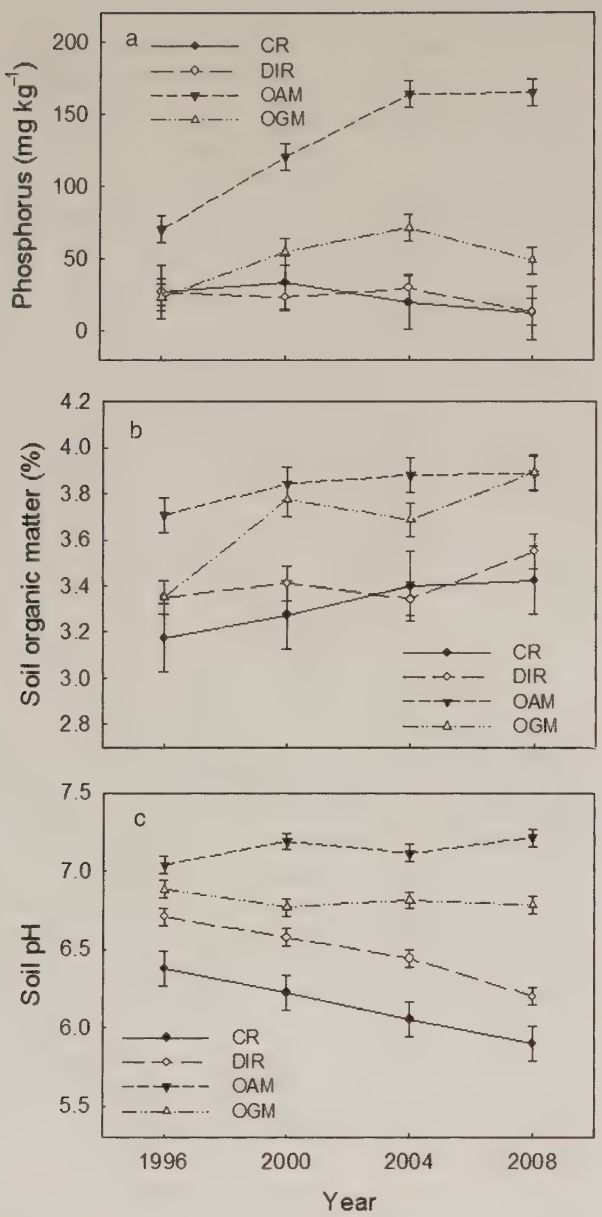


**Figure 1.** Soil (a) P (Bray-1 P), (b) soil OMC (%), and (c) pH sampled to a depth of 15 cm in conventional (CR), diversified conventional (DIR), organic animal manure-based (OAM), and organic forage-based (OFG) systems between 1996 and 2008 from the Long-Term Crop Rotation experiment near Mead, NE. Bars represent the standard error of the mean.

within systems and compared using orthogonal contrasts. A significance level of $\alpha = 0.05$ was chosen to indicate the statistical difference in all analyses.

## Results and Discussion

### *Soil properties*

Measurements of soil pH, OMC, P and K taken between 1979 and 1990 were reported by Lesoing[7]. After four cycles of the rotation (measured in the fall of 1990), soil OMC was greater in the organic treatment (3.84%) compared with the average of the three conventional treatments (3.27%), and P and K values in the organic treatment were 4.5 and 1.6 times greater, respectively, than the average of the three conventional treatments after 16 years since the field came out of uniform alfalfa[7]. These values are reported for experimental site historical reference and could not be statistically analyzed.

After 1996, many soil chemical and physical properties were affected by nutrient source and time. In 2004 and 2008, P concentrations were greater in soils receiving manure amendments (OAM and OFG) compared with soils receiving synthetic P fertilizer (CR and DIR) (Fig. 1a). Soils in the OAM system, which have received biennial manure amendments since 1975, had greater P concentrations than other systems between 1996 and 2008, confirming trends established in the first four cycles[7]. When comparing the 1996 and 2008 soil analyses, P concentrations remained stable in the CR and DIR systems, whereas concentrations increased in the OAM system (70–165 mg P kg$^{-1}$). The P levels in the OFG system were not different when comparing 1996 and 2008 results, but P levels did increase in this system in both 2000 and 2004, relative to 1996 levels. While 2008 P concentrations within the CR, DIR and OFG systems did not change relative to their respective 1996 levels, the gap between the conventional systems (slight reduction in P) and OFG system (slight increase in P) widened enough that by 2004 and 2008 P concentrations were statistically greater in the OFG system. Similarly, OMC was greater in soils with animal manure amendments (OAM and OFG; Fig. 1b). While OMC was relatively stable in the CR, DIR and OAM systems, the OFG system experienced an increase in OMC from 1996 to 2008 (3.35–3.89%). Across all years, soil pH was greatest in the OAM system, followed by OFG, DIR and CR. Over time, soils amended with synthetic fertilizers became more acidic (CR soils decreased from pH 6.38 in 1996 to 5.90 in 2008; DIR soils decreased from pH 6.71 in 1996 to 6.20 in 2008), while soils amended with manure became more alkaline (OAM soils increased from pH 7.04 in 1996 to 7.21 in 2008) or remained stable (OFG soil pH was 6.88 in 1996 and 6.78 in 2008) (Fig. 1c). The increased levels of soil P, OMC and pH in the OAM system support hypothesis 1.

In further support of hypothesis 1, soil levels of Ca, K, Mg and Zn were generally greater in cropping systems amended with animal manure between 1996 and 2008 (OAM and OFG; Table 4). Calcium levels were greater in manure-amended systems compared with synthetically amended systems in each of the four sampling years. DIR was the only system that experienced decreased levels of Ca, whereas the other three systems remained stable between 1996 and 2008. K levels were consistently greatest in the OAM system in each of the four sampling years. Soils in the OFG system contained the second-greatest K levels between 2000 and 2008. K levels increased between 1996 and 2008 in both the OAM and OFG systems, while concentrations remained stable in plots amended with synthetic fertilizers. Similarly, Mg levels were greatest in

**Table 4.** Soil fertility measurements taken between 1996 and 2008 from stage two of the Long-Term Crop Rotation experiment near Mead, NE. All soil samples were collected to a depth of 15 cm. CR, conventional; DIR, diversified conventional; OAM, organic animal manure-based; OFG, organic forage-based. Different letters following values within a given year and soil nutrient indicate a significant difference ($P < 0.05$).

| | Year | | | |
|---|---|---|---|---|
| **Soil property** | **1996** | **2000** | **2004** | **2008** |
| | ----------cmol kg$^{-1}$---------- | | | |
| Calcium | | | | |
| CR | 11.04 b | 11.28 b | 9.96 b | 9.80 b |
| DIR | 12.18 b | 12.48 b | 11.36 b | 11.13 b |
| OAM | 13.21 a | 13.72 a | 12.80 a | 13.05 a |
| OFG | 13.26 a | 13.70 a | 13.03 a | 12.69 a |
| Potassium | | | | |
| CR | 0.90 b | 0.90 c | 1.08 bc | 0.97 bc |
| DIR | 0.95 b | 1.03 c | 1.05 c | 0.95 c |
| OAM | 1.41 a | 1.76 a | 2.01 a | 1.74 a |
| OFG | 0.91 b | 1.32 b | 1.24 b | 1.23 b |
| Magnesium | | | | |
| CR | 2.98 abc | 2.73 b | 2.59 b | 2.69 b |
| DIR | 2.96 bc | 2.99 b | 2.83 b | 2.90 b |
| OAM | 3.38 a | 3.63 a | 3.53 a | 3.62 a |
| OFG | 3.33 ab | 3.43 a | 3.45 a | 3.48 a |
| Sodium | | | | |
| CR | 0.21 a | 0.09 a | 0.08 b | 0.09 a |
| DIR | 0.15 b | 0.09 a | 0.10 b | 0.10 a |
| OAM | 0.13 b | 0.11 a | 0.14 a | 0.11 a |
| OFG | 0.16 ab | 0.10 a | 0.12 ab | 0.10 a |
| | ----------mg kg$^{-1}$---------- | | | |
| Sulfate | | | | |
| CR | 10.50 a | 11.75 b | 12.00 b | 12.50 a |
| DIR | 10.50 a | 10.94 b | 11.19 b | 11.38 a |
| OAM | 11.69 a | 15.31 a | 16.25 a | 12.44 a |
| OFG | 11.31 a | 15.06 a | 13.56 b | 12.56 a |
| Zinc | | | | |
| CR | 0.85 ab | 1.02 b | 1.04 c | 0.88 bc |
| DIR | 0.85 b | 1.58 b | 0.97 c | 0.75 c |
| OAM | 1.87 a | 2.71 a | 3.83 a | 3.24 a |
| OFG | 0.79 b | 1.42 b | 2.19 b | 1.50 b |

the organic systems between 2000 and 2008. In 1996, Mg levels were greatest in the OAM system. Between 1996 and 2008, Mg levels remained stable in all systems. Zn levels were greatest in the OAM system between 2000 and 2008. In 1996, levels of Zn in the OAM system were greater than levels in the DIR and OFG system, but not the CR system. Between 1996 and 2008, Zn levels increased in the organic systems and remained stable in the conventional systems (CR and DIR). Soil levels of Na and sulfate were inconsistent among systems and over time. However, Na levels generally decreased across all treatments between 1996 and 2008.

Soil fertility in organic production systems is maintained with organic amendments such as farmyard manure and including forage legumes (e.g., alfalfa) and green manures in the crop rotation[27]. In this long-term experiment, differences in soil properties were likely due to differences in crop sequence, soil amendments, frequency of tillage and nutrient inputs among cropping systems (Tables 5–7). Phosphorus, OMC, pH, Ca, K, Mg and Zn were all greater in soils amended with manure (OAM and OFG) compared to soil amended with synthetic fertilizers (CR and DIR). Greater concentrations of P, OMC, Ca, K and Mg in organically amended soils have been observed in many previous studies[15,16,27,30]. However, despite the lack of K inputs and subsequent negative nutrient budgets in both the CR and DIR systems ($-0.53$ and $-0.41$ cmol K kg$^{-1}$, respectively), K levels did not change in the conventional systems between 1996 and 2008 (Table 5). Indeed, K fertilizer application in eastern Nebraska soils is rare and in some cases can result in grain yield reductions[31]. In most situations, sufficient background soil levels combined with K supplied from the organic matter fraction is sufficient for the sustainable maintenance of long-term cropping systems in eastern Nebraska[32]. The general reduction of Na over time in organically amended soils of this study is unique, given that the application of animal manure often leads to salinity problems in soil[33]. This reduction in Na may be attributed to the presence of sorghum in all systems since 1996, as sorghum has demonstrated high Na removal efficiency and can be used in the reclamation of sodic soils[34].

It is interesting to note that soil P levels remained relatively stable in the OFG system between 1996 and 2008, despite a substantial surplus in the P nutrient budget for this system (Table 6). Indeed, at least two previous studies have demonstrated a negative P balance when alfalfa is included in rotation[18,19]. However, another study found that continuous alfalfa production increased the soil organic P fraction compared with grain-based annual cropping systems[35]. It is possible that stable P is extracted from the soil by alfalfa roots and transformed into a more labile form upon root death[35]. Moreover, alfalfa serves as a host to arbuscular mycorrhizal fungi, which have been shown to increase the availability and uptake of soil P[36]. Despite the potential for increased soil P availability associated with alfalfa production, the removal of P in the form of multiple hay cuttings contributed to relatively low levels of soil P in the OFG system, considering the surplus of P inputs via animal manure (Table 6). As expected, frequent manure application to meet grain crop N demands resulted in a substantial surplus of soil P in the OAM system (Table 6 and Fig. 1a). Similar to K, soil P levels did not change between 1996 and 2008 in the conventional systems despite infrequent fertilizer application and subsequently negative nutrient balances ($-111$ mg P kg$^{-1}$ in the CR system and $-58$ mg P kg$^{-1}$ in the DIR system; Table 6). While background levels of P are not typically as high as background levels of K in eastern Nebraska soils; the release of P from labile organic matter and microbial biomass appeared to compensate for the negative P balances in the conventional systems[32].

**Table 5.** Farm gate nutrient budget to account for K inputs and outputs within and among cropping systems between 1996 and 2007 in the Long-Term Crop Rotation experiment near Mead, NE. Each crop phase was treated as an equal component (3 years) of the 12-year study period in this budget (e.g., manure was applied 12 times in all OAM corn experimental units between 1996 and 2007, but corn would only be present in 3 of 12 years in a given experimental unit; thus, the total number of applications was divided by 4). OAM, organic animal manure-based; OFG, organic forage-based; CR, conventional; DIR, diversified conventional.

| System and crop | Inputs | K content of inputs (%) | Input rate ($kg\,ha^{-1}$) | No. of applications | Total K input ($kg\,ha^{-1}$) | K content of crop[1] (%) | Crop yield ($kg\,ha^{-1}$) | Total K removed[2] ($kg\,ha^{-1}$) | Total K balance | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | ($kg\,ha^{-1}$) | ($cmol\,kg^{-1}$) |
| OAM soybean | None | | | | | 1.542 | 1990 | 92 | −92 | −0.12 |
| OAM corn | Manure | 2.46 | 31,700 | 3 | 2339 | 0.344 | 6558 | 68 | 2272 | 3.03 |
| OAM soybean | None | | | | | 1.542 | 1990 | 92 | −92 | −0.12 |
| OAM wheat | Manure | 2.07 | 25,400 | 2 | 1052 | 0.487 | 3151 | 46 | 1006 | 1.34 |
| **OAM total** | | | | | **3391** | | | **298** | **3093** | **4.13** |
| OFG alfalfa | Manure | 2.54 | 41,700 | 1.5 | 1589 | 2.119 | 7413 | 471 | 1118 | 1.49 |
| OFG alfalfa | None | | | | | 2.119 | 7413 | 471 | −471 | −0.63 |
| OFG corn | Manure | 4 | 32,400 | 0.25 | 324 | 0.344 | 5045 | 52 | 272 | 0.36 |
| OFG wheat | None | | | | | 0.487 | 2487 | 36 | −36 | −0.05 |
| **OFG total** | | | | | **1913** | | | **1031** | **882** | **1.18** |
| CR corn | None | | | | | 0.344 | 7379 | 76 | −76 | −0.10 |
| CR soybean | None | | | | | 1.542 | 2597 | 120 | −120 | −0.16 |
| CR sorghum | None | | | | | 0.415 | 6364 | 79 | −79 | −0.11 |
| CR soybean | None | | | | | 1.542 | 2597 | 120 | −120 | −0.16 |
| **CR total** | | | | | | | | **396** | **−396** | **−0.53** |
| DIR corn | None | | | | | 0.344 | 7651 | 79 | −79 | −0.11 |
| DIR sorghum | None | | | | | 0.415 | 6138 | 76 | −76 | −0.10 |
| DIR soybean | None | | | | | 1.542 | 2369 | 110 | −110 | −0.15 |
| DIR wheat | None | | | | | 0.487 | 2870 | 42 | −42 | −0.06 |
| **DIR total** | | | | | | | | **307** | **−307** | **−0.41** |

[1] NRCS Plant Nutrient Content Database[28].
[2] Total K removed = (% K content of grain or forage) × (grain or forage yield) × (3 years in rotation per experimental unit).

In this study, we observed greater pH in soils receiving organic amendments compared with soils receiving synthetic fertilizers, which is consistent with the results of several previous studies[16,37,38]. Moreover, the reduction in soil pH in the conventional systems over time is congruent with the results of another long-term cropping systems experiment conducted at the ARDC near Mead, NE where soil pH decreased as synthetic N rate increased[39]. In a similar long-term rotation study, the pH of soils in organic and 'zero-input' rotations was greater than pH levels in a 'high-input' corn–soybean rotation[2]. Addition of organic amendments may raise the pH of soils by complexing aluminum and increasing base saturation[40,41]. The calcium carbonate excreted in animal manure serves as a liming source in soils, often raising soil pH. When manure is applied in high quantities to meet crop N demand soil pH has been shown to increase, whereas manure application to meet crop P demand has been shown to stabilize soil pH[42]. These findings are consistent with the results of this study where soil pH increased over time in the OAM system (manure applied to meet crop N demand) and remained stable in the OFG system (manure applied to meet crop P demand). The reduced soil pH over time in the conventional systems was likely the result of synthetic fertilizer application. While few synthetic fertilizers are actually acidic, many are acid forming. This soil acidification is most often caused by microbial oxidation of ammoniacal fertilizers resulting in nitric acid accumulation. Much of the excess nitric acid is then stored in organic matter and the soil pH decreases[43].

The results of this study are in contrast to those found in several European experiments that have demonstrated negative nutrient balances and decreasing pH values in organic cropping systems. For example, Eltun et al.[44] reported an increasingly negative nutrient balance for P and K in organic arable and forage cropping systems, despite organic amendments. Moreover, Breland and Eltun[45] concluded that arable cropping systems (dominated by annual cereals) have poorer soil fertility than forage-based systems. The OFG system in this study included 2 years of forage within each 4-year cycle, but the nutrient levels in the OAM system were still greater than those in the OFG system. Kirchmann et al.[4] and Gosling and Shephard[9] reported lower pH values in organically managed soils compared with those amended with synthetic fertilizer. Crop rotations that rely heavily on legumes for symbiotic nitrogen fixation may acidify soils because legumes release more protons from their roots than non-legume cereal crops[46]. Both organic rotations in this study included a leguminous crop (soybean or alfalfa) in 2 of the 4 years of each cycle; however, soil pH remained stable (OFG) or increased (OAM) between 1996 and 2008. Contradictory

**Table 6.** Farm gate nutrient budget to account for phosphorus inputs and outputs within and among cropping systems between 1996 and 2007 in the Long-Term Crop Rotation experiment near Mead, NE. Each crop phase was treated as an equal component (3 years) of the 12-year study period in this budget. OAM, organic animal manure-based; OFG, organic forage-based; CR, conventional; DIR, diversified conventional.

| System and crop | Inputs | P content of inputs (%) | Input rate (kg ha$^{-1}$) | No. of applications | Total P input (kg ha$^{-1}$) | P content of crop[1] (%) | Crop yield (kg ha$^{-1}$) | Total P removed[2] (kg ha$^{-1}$) | Total P balance (kg ha$^{-1}$) | Total P balance (mg kg$^{-1}$) |
|---|---|---|---|---|---|---|---|---|---|---|
| OAM soybean | None | | | | | 0.659 | 1990 | 39 | −39 | −20 |
| OAM corn | Manure | 0.693 | 31,700 | 3 | 659 | 0.317 | 6558 | 62 | 597 | 311 |
| OAM soybean | None | | | | | 0.659 | 1990 | 39 | −39 | −20 |
| OAM wheat | Manure | 0.595 | 25,400 | 2 | 302 | 0.435 | 3151 | 41 | 261 | 136 |
| **OAM total** | | | | | **961** | | | **182** | **779** | **406** |
| OFG alfalfa | Manure | 0.737 | 41,700 | 1.5 | 461 | 0.261 | 7413 | 58 | 403 | 210 |
| OFG alfalfa | None | | | | | 0.261 | 7413 | 58 | −58 | −30 |
| OFG corn | Manure | 0.77 | 32,400 | 0.25 | 62 | 0.317 | 5045 | 48 | 14 | 7 |
| OFG wheat | None | | | | | 0.435 | 2487 | 32 | −32 | −17 |
| **OFG total** | | | | | **523** | | | **197** | **327** | **170** |
| CR corn | Phosphate | 46 | 40 | 3 | 13.8 | 0.317 | 7379 | 70 | −56 | −29 |
| CR soybean | None | | | | 0 | 0.659 | 2597 | 51 | −51 | −27 |
| CR sorghum | Phosphate | 46 | 40 | 3 | 13.8 | 0.352 | 6364 | 67 | −53 | −28 |
| CR soybean | None | | | | 0 | 0.659 | 2597 | 51 | −51 | −27 |
| **CR total** | | | | | **27.6** | | | **240** | **−212** | **−111** |
| DIR corn | Phosphate | 46 | 40 | 0.75 | 13.8 | 0.317 | 7651 | 73 | −59 | −31 |
| DIR sorghum | Phosphate | 46 | 40 | 0.75 | 13.8 | 0.352 | 6138 | 65 | −51 | −27 |
| DIR soybean | None | | | | | 0.659 | 2369 | 47 | −47 | −24 |
| DIR wheat | Phosphate | 46 | 60 | 3 | 82.8 | 0.435 | 2870 | 37 | 45 | 24 |
| **DIR total** | | | | | **110.4** | | | **222** | **−111** | **−58** |

[1] NRCS Plant Nutrient Content Database[28].
[2] Total P removed = (% P content of grain or forage) × (grain or forage yield) × (3 years in rotation per experimental unit).

results on this issue suggest the continued importance of monitoring soil pH and mineral nutrients in conventional and organic systems. Organic farmers face greater limitations when dealing with soil mineral deficiencies since they cannot apply synthetic fertilizers; thus, the maintenance of soil fertility through system design is essential to the long-term sustainability of organic cropping systems[44].

In a 21-year long-term trial, nutrient inputs were 34–51% lower in organic systems compared with conventional systems, while yields in organic systems were only 20% lower[37]. This may suggest crops in organic systems utilize nutrients more efficiently than the same crops in conventional systems. An apparent increase in nutrient-use efficiency may be the result of increased mineralization of OMC in organic cropping systems. Moreover, soil OMC is widely recognized as a key indicator of soil quality[47]. Total organic carbon concentration (TOC), a measure of OMC, increased by 9% in the top 15 cm of the soil after 4 years of organic management in Iowa, while TOC in conventionally managed plots increased by only 3%[27]. In a summary of long-term trials across the US, both 'legume-only' and 'legume + animal manure' organic systems increased soil organic carbon (SOC) by 14% compared with conventional systems after an average of 10 years[14]. Similarly, Kirchmann et al.[4] reported greater SOC in organic systems compared with conventional. However, this was a result of 40% greater carbon inputs in the organic system; the organic system included solid manure, cover crops and greater weed residue for incorporation[4]. Therefore, the greater levels of OMC observed in the OAM and OFG systems are likely due to the greater amount of carbon inputs (e.g., bovine manure, alfalfa, weed residue; Table 7). Regardless of carbon inputs, the results of this study are congruent with previous findings in that OMC was 10% greater in the organic systems compared to the conventional systems as of 2008. However, in the absence of soil bulk density data, we cannot eliminate the possibility that differences in OMC among systems were due in part to differences among cropping systems in soil bulk density. Indeed, the variable management strategies (e.g., tillage intensity and manure application) associated with each system may have led to changes in soil bulk density[48].

One study revealed greater levels of total soil carbon in organic legume systems (without animal manure inputs) compared to conventional systems after 20+ years of arable farming[12]. This is significant because estimates of net annual soil carbon inputs were similar for the two systems[3]. Drinkwater et al.[12] demonstrated that the use of low C : N residue (e.g., legumes) to maintain soil fertility, coupled with greater temporal diversity of crop rotations, can increase the retention of soil carbon. Therefore, the soil OMC in the organic systems of this study may have increased over time due to a combination of increased carbon inputs and the inclusion of low C : N crop residue.

**Table 7.** Carbon inputs within and among cropping systems between 1996 and 2007 in the Long-Term Crop Rotation experiment near Mead, NE. Each crop phase was treated as an equal component (3 years) of the 12-year study period in all calculations. OAM, organic animal manure-based; OFG, organic forage-based; CR, conventional; DIR, diversified conventional.

| System and crop | Inputs | Manure C content (%) | Manure rate[1] (kg ha$^{-1}$) | Crop residue C content[2] (%) | Crop yield (kg ha$^{-1}$) | Harvest index[2] (%) | Weed input[3] (kg ha$^{-1}$) | Weed C content[2] (%) | Total C input[4] (kg ha$^{-1}$) |
|---|---|---|---|---|---|---|---|---|---|
| OAM soybean | None | | | 15 | 1990 | 46 | 2024 | 25 | 2002 |
| OAM corn | Manure | 10 | 31,700 | 30 | 6558 | 52 | 2488 | 25 | 33,229 |
| OAM soybean | None | | | 15 | 1990 | 46 | 2024 | 25 | 2002 |
| OAM wheat | Manure | 10 | 25,400 | 50 | 3151 | 51 | 192 | 25 | 17,700 |
| **OAM total** | | | | | | | | | **54,932** |
| OFG alfalfa | Manure | 10 | 41,700 | 32.5 | 7413 | 100 | 0 | 25 | 18,765 |
| OFG alfalfa | None | | | 32.5 | 7413 | 86.5 | 0 | 25 | 976 |
| OFG corn | Manure | 10 | 32,400 | 30 | 5045 | 52 | 3715 | 25 | 7396 |
| OFG wheat | None | | | 50 | 2487 | 51 | 729 | 25 | 2375 |
| **OFG total** | | | | | | | | | **29,511** |
| CR corn | None | | | 30 | 7379 | 52 | 64 | 25 | 3236 |
| CR soybean | None | | | 15 | 2597 | 46 | 93 | 25 | 701 |
| CR sorghum | None | | | 40 | 6364 | 49 | 188 | 25 | 4036 |
| CR soybean | None | | | 15 | 2597 | 46 | 93 | 25 | 701 |
| **CR total** | | | | | | | | | **8673** |
| DIR corn | None | | | 30 | 7651 | 52 | 291 | 25 | 3523 |
| DIR Sorghum | None | | | 40 | 6138 | 49 | 233 | 25 | 3931 |
| DIR soybean | None | | | 15 | 2369 | 46 | 15 | 25 | 587 |
| DIR wheat | None | | | 50 | 2870 | 51 | 216 | 25 | 2271 |
| **DIR total** | | | | | | | | | **10,313** |

[1] Total manure C input based on manure rate and application frequency data from Tables 5 and 6.
[2] Assumed values.
[3] Aboveground weed biomass (Wortman et al.[29]).
[4] Total C input = ((((% manure C content) × (manure rate) × (no. of applications per crop phase per 12 years)) + ((% crop residue C content) × (grain or forage yield) × (1 − (% harvest index))) + ((% weed residue C content) × (weed residue input))) × (3 years in rotation per experimental unit).

The rotation in the OAM system included 2 years of soybean and the rotation in the OFG system included 2 years of alfalfa (both legumes with low C : N ratios). In another study, OMC decreased an average of 2% annually in an annual crop/plowed system, indicating the importance of forage crops for the maintenance of OMC[17]. Including forage crops in the rotation (at least 50%, as in the OFG system) can also increase soil stability, soil porosity and available soil water[17]. The soil-building benefits of forage crops combined with the reduction of tillage during this crop phase may help explain the similar levels of soil OMC in the OAM and OFG systems, despite fewer carbon inputs in the OFG system. In 2008, OMC was not different between the OAM and OFG systems despite approximately 46% greater carbon inputs in the OAM system (Table 7). Tillage consistently reduces levels of soil OMC compared to no-till practices[49]; thus, the lack of tillage during the 2-year alfalfa phase in the OFG system may have negated the greater carbon inputs in the OAM system with regard to building soil OMC.

Overall, it seems that manure application in the organic systems is more effective in building soil P than OMC. Manure application rates in the OAM system were based on subsequent crop N demands, which often results in the accumulation of soil P because the N : P ratio of the manure source is usually smaller than the N : P ratio of crop uptake, as was the case in this study (Table 2)[50]. Moreover, while manure is a substantial source of organic carbon and should increase soil OMC, the extensive use of tillage in the organic systems will result in the accelerated microbial oxidation of this carbon source[51].

While it may not be possible to determine mechanistic differences among systems due to varying management practices between the conventional and organic systems (e.g., manure application in the organic systems), it is clear that organic cropping systems with manure amendments appear to increase soil nutrients, soil pH and OMC. Moreover, despite frequent tillage operations organic systems tend to retain greater amounts of soil organic matter[14]. The demonstrated ability of both animal manure- and legume-based organic systems to build soil OMC and maintain soil nutrients indicates great potential for the long-term sustainability of organic farming.

## *Crop yields*

The average daily temperatures during the growing season for summer annual crops (April 1–September 30) and

**Table 8.** Analysis of variance for the effects of precipitation, system and system × precipitation on yield of each crop between 1996 and 2007 from the Long-Term Crop Rotation experiment near Mead, NE. For corn, sorghum and soybean precipitation values were summed between April 1 and September 30, and for wheat precipitation values were summed between October 1 and June 15.

| Crop | Parameter | *P*-value |
|---|---|---|
| Corn | System, S | < 0.0001 |
| | Precipitation, P | < 0.0001 |
| | S × P | 0.2626 |
| Sorghum | System, S | < 0.0001 |
| | Precipitation, P | 0.0059 |
| | S × P | 0.6079 |
| Soybean | System, S | < 0.0001 |
| | Precipitation, P | < 0.0001 |
| | S × P | 0.0608 |
| Wheat | System, S | < 0.0001 |
| | Precipitation, P | 0.0039 |
| | S × P | 0.4843 |

winter wheat (October 1–June 15) between 1996 and 2007 were 19.2 and 6.5°C, respectively (Table 3). These values were warmer than the 30-year mean temperatures (1971–2000) for Mead, NE, which were 19.0 and 5.8°C for summer annual and winter wheat-growing seasons, respectively. The average total precipitation values during the growing season for summer annual crops and winter wheat between 1996 and 2007 were 471 and 397 mm. This was drier than the 30-year mean precipitation values (1971–2000) for summer annual and winter wheat-growing seasons, which were 519 and 411 mm, respectively (Table 3).

Analysis of yield data between 1996 and 2007 indicated a year by cropping system interaction effect for all crops. Annual climatic variation did little to explain these interactions (Table 8); thus, yield data reported here have been averaged across all study years to better understand yield response to the long-term management associated with each cropping system. However, annual yields for each grain crop are presented in Figure 2 to demonstrate this interaction.

From 1996 to 2007, average corn yields were greatest in the DIR (7.65 Mg ha$^{-1}$) and CR systems (7.38 Mg ha$^{-1}$), followed by the OAM (6.56 Mg ha$^{-1}$) and OFG (5.05 Mg ha$^{-1}$) systems (Fig. 2). In accordance with hypotheses 2 and 3, yields were higher in the OAM system compared with the OFG system, and among all systems yields were greatest in the conventional systems. Both growing season precipitation (April 1–September 30) and system affected corn yield, but in contrast to hypothesis 4 there was no precipitation × system interaction (Table 8). In contrast to hypothesis 5, neither P nor OMC affected corn yield, but there was a significant P × system interaction (Table 9).

From 1996 to 2007, average sorghum yield was greatest in the CR (6.36 Mg ha$^{-1}$) and DIR systems (6.14 Mg ha$^{-1}$), followed by the OAM (5.25 Mg ha$^{-1}$) and OFG (4.59 Mg ha$^{-1}$) systems (Fig. 2). In accordance with hypotheses 2 and 3, yield was greater in the OAM system compared with the OFG system and among all systems yields were highest in the conventional systems. Both precipitation (April 1–September 30) and system affected sorghum yield, but in contrast to hypothesis 4 there was no precipitation × system interaction (Table 8). In contrast to hypothesis 5, neither P nor OMC affected sorghum yields (Table 9).

From 1996 to 2007, average soybean yield was highest in the CR (2.60 Mg ha$^{-1}$) and DIR systems (2.37 Mg ha$^{-1}$) and lowest in the OAM (1.99 Mg ha$^{-1}$) system (Fig. 2). In accordance with hypothesis 3, yield was greater in the conventional systems compared with the OAM system. Precipitation (April 1–September 30) affected soybean yield, but in contrast to hypothesis 4 there was no precipitation × system interaction (Table 8). OMC did affect soybean yield; however, yield decreased as organic matter increased, which is not consistent with hypothesis 5. Soil P levels did not affect soybean yield (Table 9).

From 1996 to 2007, average wheat yield was greatest in the OAM system (3.15 Mg ha$^{-1}$), followed by DIR (2.87 Mg ha$^{-1}$) and OFG (2.49 Mg ha$^{-1}$) systems (Fig. 2). In accordance with hypothesis 2, yield was greater in the OAM system compared to the OFG system. However, among all systems, yield was greatest in the OAM system which contradicts hypothesis 3. Precipitation (October 1–June 15) affected wheat yield, but in contrast to hypothesis 4 there was no precipitation × system interaction (Table 8). In support of hypothesis 5, soil P levels did affect wheat yields (Table 9). Wheat yields were greatest in the OAM system, which also had the greatest soil P levels. However, in contrast to hypothesis 5, OMC did not affect wheat yields (Table 9).

From 1996 to 2007, average alfalfa forage yields were 7.41 Mg ha$^{-1}$ ± 0.33 Mg ha$^{-1}$ (mean ± one standard error) in the OFG system. This is the only system that included alfalfa therefore yields were compared with the same 12-year county average for conventional alfalfa forage production. The Saunders County, NE (location of ARDC) average during this period for non-irrigated alfalfa hay was 7.64 Mg ha$^{-1}$, which is not substantially greater than the 12-year yield mean in the OFG system as it falls within one standard error[21].

Several recent studies have demonstrated the competitiveness of long-term organic cropping systems compared with conventional[2,3,6]. While the results of these studies vary by year, crop and climate, the accumulation and comparison of data from around the US will aid in the design of appropriate organic and conventional cropping systems across a diverse range of ecoregions.

Corn is among the most important agronomic crops in the USA; thus, it is commonly included in comparative studies between organic and conventional crop production. In a long-term trial at the Rodale Institute in Pennsylvania, corn grain yields did not differ among organic animal manure-based, organic legume-based and conventional

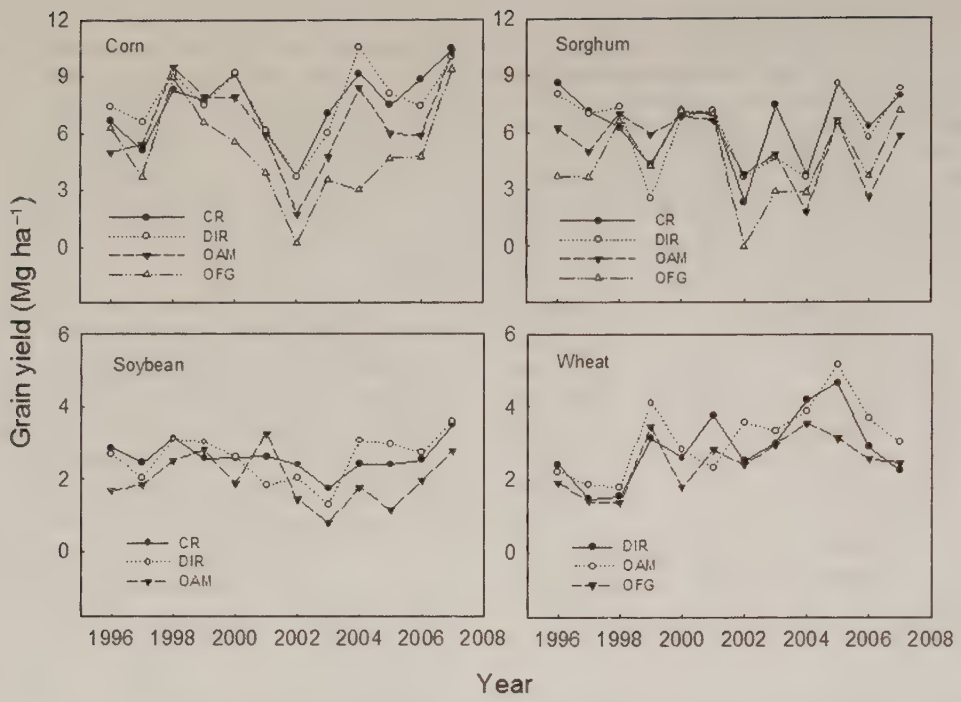


**Figure 2.** Grain yields for corn, sorghum, soybean and winter wheat in conventional (CR), diversified conventional (DIR), organic animal manure-based (OAM) and organic forage-based (OFG) systems between 1996 and 2007 in the Long-Term Crop Rotation experiment near Mead, NE. Standard errors for annual corn yields were CR ± 0.76 Mg ha$^{-1}$; DIR, OAM and OGM ± 0.54 Mg ha$^{-1}$. Standard errors for annual sorghum yields were CR ± 0.55 Mg ha$^{-1}$; DIR, OAM and OGM ± 0.39 Mg ha$^{-1}$. Standard errors for annual soybean yields were CR ± 0.44 Mg ha$^{-1}$; DIR ± 0.31 Mg ha$^{-1}$ and OAM ± 0.22 Mg ha$^{-1}$. Standard errors for annual wheat yields were DIR, OAM and OGM ± 0.22 Mg ha$^{-1}$.

production systems[3]. Similarly, Posner et al.[6] and Porter et al.[2] reported organic corn grain yields 91–93% of the conventional system yields in their respective studies. In this study, compared to the average of the two conventional systems (CR and DIR), the OAM system yielded 87% of the conventional corn yields. Less comparable corn yields in organic systems have also been reported. Cavigelli et al.[5] observed organic corn grain yields to be 76% of conventional yields. These results are similar to corn in the OFG system that yielded 67% of the two conventional systems. Sorghum responded similarly to the different management systems; compared with the average of two conventional systems the OAM system yielded 84% and the OFG system yielded 73%. The greater sorghum yields in the OAM system compared to the OFG system are similar to the results of Mady Kaye et al.[10], who observed an 8% increase in sorghum yields when amended with manure and rotated with soybean.

Organic soybean grain yields were much less competitive in this study compared with the results of two recent studies. Posner et al.[6] reported organic soybean yields at 92% of the conventional yields and Pimentel et al.[3] found no difference between organic and conventional soybean yields over a 20-year period (excluding one extreme drought year). In contrast, two other recent studies have reported organic soybean yields in four-year rotations that yielded 81–84% compared with the conventional systems[2,5]. These results are similar to yields observed in this study, where organic (OAM) soybean yields were only 80% of the average of the two conventional system yields. Early season, wet field conditions that limit mechanical weed control, especially rotary hoeing, in organic systems can be particularly damaging to soybean yields[29]. Porter et al.[2] observed organic soybean yields that were only 58 and 69% of conventional yields in unusually wet years, but organic yields between 96 and 99% of conventional yields in 'normal' precipitation years. These results expose the vulnerability of organic cropping systems due to heavy reliance on timely mechanical weed control.

Organic wheat yields seem to be far more competitive with conventional yields than organic corn, sorghum or soybean yields. Organic wheat yields reported by Posner et al.[6] did not differ from the conventional county average. Similarly, organic wheat yields reported by Mader et al.[37] were 90% of conventional yields, and wheat yields reported by Cavigelli et al.[5] were not different between conventional and organic systems. Compared with the conventional system (DIR), we observed greater yields in the OAM system (110%), but reduced yields in the OFG system (87%). Manure application has been identified as an important contributor to improved yields in wheat cropping systems, which may explain the significant yield increase in the OAM system[5,52]. These results suggest winter wheat may be a competitive crop in organic cropping systems in Nebraska, and thus a logical crop choice for organic farmers as an important component of their rotations.

**Table 9.** Analysis of variance for the effect of system, organic matter content (OMC), phosphorus (P), system × OMC and system × P on yield of each crop in the first crop phase between 1996 and 2004 (1996, 2000 and 2004) from the Long-Term Crop Rotation experiment near Mead, NE.

| Crop | Parameter | *P*-value |
|---|---|---|
| Corn | System, S | < 0.0001 |
| | Organic matter, OMC | 0.7775 |
| | Phosphorus, P | 0.2988 |
| | S × OMC | 0.2583 |
| | S × P | 0.0477 |
| Sorghum | System, S | 0.0008 |
| | Organic matter, OMC | 0.4581 |
| | Phosphorus, P | 0.9488 |
| | S × OMC | 0.9527 |
| | S × P | 0.4193 |
| Soybean | System, S | 0.0157 |
| | Organic matter, OMC | 0.0012 |
| | Phosphorus, P | 0.4672 |
| | S × OMC | 0.9993 |
| | S × P | 0.1241 |
| Wheat | System, S | < 0.0001 |
| | Organic matter, OMC | 0.7322 |
| | Phosphorus, P | 0.0012 |
| | S × OMC | 0.8762 |
| | S × P | 0.9668 |

Alfalfa may also be a logical crop choice for organic farmers. In addition to the many soil biological, chemical and physical benefits of growing forages in rotation with grain crops, it seems organic alfalfa yields are also highly competitive with conventional alfalfa yields[2,38]. While alfalfa was only present in the OFG system of this experiment, when compared with the Saunders County, NE conventional average between 1996 and 2007, there was no statistical difference (i.e., the 12-year mean of OFG alfalfa yield ± one standard error includes the 12-year mean of county alfalfa yields). Conventional producers often manage alfalfa with minimal inputs, but growing alfalfa on certified organic land will allow farmers to market the forage as a certified organic product, increasing the profitability of including alfalfa in rotation. However, many farmers are reticent to export a forage crop such as alfalfa from the farm, and prefer to feed livestock to add value to this important resource and minimize export of valuable nutrients from the system.

Many of the yield reductions in the organic systems seem to be the result of increased weed cover and biomass. Porter et al.[2] and Posner et al.[6] reported that in years when mechanical weed control was effective, organic grain yields were equivalent to conventional yields. In effective weed control years, organic corn and soybean yields were 90–98% of conventional yields, but when weed control was not effective organic corn and soybean yields were only 69–80% of conventional yields[6]. These results suggest weed pressure was the major limiting factor in organic corn and soybean production. In 2007 and 2008, we observed greater broadleaf and grass weed biomass in corn and soybean plots of the organic systems compared with the conventional systems[29]. In the absence of nutrient deficiencies, the increased weed biomass is likely responsible for the yield loss observed within the organic systems of this study. While grain yields were generally reduced in the organic systems compared with the conventional systems, lower production costs may increase the economic competitiveness of organic systems even without considering current organic price premiums[2,3]. However, in this long-term study, production costs were similar between conventional and organic systems (data not shown).

**Table 10.** Critical soil test values for crops grown in the Long-Term Crop Rotation experiment near Mead, NE according to Ferguson[23].

| | Crop | | | | |
|---|---|---|---|---|---|
| Soil property | Corn | Sorghum | Soybean | Wheat | Alfalfa |
| pH | 5.1–7.6 | 5.1–7.7 | 5.1–7.6 | 5.1–7.9 | 5.9–7.9 |
| P ($mg\,kg^{-1}$) | 15 | 15 | 10 | 15–25 | 25 |
| K ($cmol\,kg^{-1}$) | 0.32 | 0.32 | 0.32 | 0.32 | 0.32 |
| Mg ($cmol\,kg^{-1}$) | 0.42 | 0.42 | 0.42 | 0.42 | 0.42 |
| $SO_4$ ($mg\,kg^{-1}$) | 6 | 6 | – | – | 5 |
| Zn ($mg\,kg^{-1}$) | 0.8 | 0.8 | 0.4 | – | – |

Overall, there was no positive effect of OMC on grain yield in any crop. It was hypothesized that increasing OMC would increase crop yield due to the capacity for increased nutrient mineralization and soil water retention[3,14,53,54]. In fact, there was a negative relationship between OMC and soybean yield, which was more likely the result of a positive relationship between OMC and weed biomass in the OAM system ($P < 0.01$; data not shown).

Similarly, there was no positive relationship between soil P and grain yield in any crop except for winter wheat. Phosphorus was maintained near sufficiency levels for corn, sorghum and soybean in all systems, which decreases the likelihood of a positive yield response. However, P sufficiency levels in wheat are greater than those for corn, sorghum and soybean[23] (Table 10). Phosphorus levels in the DIR system were between 12 and 29 $mg\,kg^{-1}$ Bray-1 P throughout the experiment, which may explain the reduced yields in this system compared with the OAM system where Bray-1 P levels were maintained well above sufficiency levels. Moreover, sufficient P nutrition in wheat has been shown to reduce water stress, which may have further contributed to greater yields in the OAM system, especially in dry years[55]. These results suggest that wheat is the most responsive crop to animal manure applications and/or P fertilization, further increasing its usefulness in organic cropping systems. In addition to these benefits, the dense plant spacing and tillering capacity of wheat often results in comparable weed suppression between organic and conventional systems[29].

While precipitation positively affected grain yields across all years and crops, in contrast to hypothesis 5 there was not a significant precipitation by system interaction. This was surprising given the varying levels of soil OMC among systems. Increased OMC is associated with greater soil water-holding capacity; thus, we expected yield to decrease less in systems with greater OMC (OAM and OFG) during years of less than average rainfall[54]. This effect was observed in Pennsylvania where corn and soybean yields increased in organic systems with greater OMC compared with conventional systems during dry years[56]. Despite differences among systems in this study, soil OMC was relatively high in all systems (>3.3%), which may explain the lack of precipitation by system interaction effect on yield. Therefore, it is unlikely that dry weather conditions negatively affected one system more than the others.

## Conclusions

The results of this study indicate that yields of corn, sorghum and soybean were greatest in conventional cropping systems, but wheat yields were greatest in an animal manure-based organic cropping system. As expected, the application of animal manure in an organic cropping system increased soil OMC, pH, P, Ca, K, Mg and Zn. However, the potential yield benefits associated with these increases were often negated by intense weed pressure in the organic system. Despite the increase in OMC due to manure application, it may be necessary to include perennial forage, such as alfalfa, in organic crop rotations to prevent losses of soil carbon due to the extensive tillage that is common practice in organic annual-cropping systems. In addition to the improved soil properties associated with alfalfa, this forage crop also provides competitive crop yields in organic compared with conventional systems (when compared with the 12-year county average for conventional alfalfa yield). Future studies should be focused toward a greater understanding of the mechanistic responses of soil biological and physical properties associated with long-term organic management. Moreover, the economics, energy efficiency and greenhouse gas emissions associated with long-term organic management represent areas of increasing interest and potential research opportunity.

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# Using reference values to assess environmental sustainability of dairy farms

I. Acosta-Alba[1,2], M.S. Corson[1,2]*, H.M.G. van der Werf[1,2], and P. Leterme[1,2]

[1]INRA, UMR1069, Soil Agro and hydroSystem, F-35000 Rennes, France.
[2]Agrocampus Ouest, F-35000 Rennes, France.
*Corresponding author: michael.corson@rennes.inra.fr

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## Abstract

One challenge for the environmental assessment of agricultural systems is to progress from estimating whether one farming system has fewer impacts than another to estimating whether or not it can be considered environmentally sustainable. To this end, we developed reference values (RVs) of farm emissions or energy use per hectare that defined hypothetical sustainability thresholds in each of three impact categories: climate change, water-quality degradation and non-renewable energy use. We applied one RV per category to the potential impacts (estimated by life-cycle assessment) of 45 French dairy farms to identify farms below RVs in each impact category and then evaluated their management and production characteristics. Seventeen of the 45 farms lie below at least one of the three RVs. Groups of farms below RVs had a higher percentage of organic farms, larger mean usable agricultural area, longer mean pasture residence time and lower mean inputs of concentrate feed and nitrogen than those above the same RVs. In consequence, the groups below RVs also tended to have lower mean milk production per cow and per hectare. All milk production systems can move toward environmental sustainability even though, according to production mode and intensity, some potential impacts are easier to reduce than others. Most farms were unable to attain the lowest RVs, suggesting that policy-makers may need to consider less ambitious RVs for existing agricultural systems. Otherwise, the distance between normative RVs and indicator values of dairy farms suggest that production and consumption of agricultural products will need to change if sustainability goals do not.

**Key words:** sustainability, reference value, life-cycle assessment, dairy

## Introduction

One challenge for the environmental assessment of agricultural systems is to progress from estimating whether one farming system has fewer impacts than another to estimating whether or not it can be considered environmentally sustainable. Although the concept of sustainability has many definitions[1–3], it has been accepted as humanity's goal to pursue since the Brundtland Commission report[4].

Sustainable development of agriculture is of great importance not only because agriculture frequently is associated with resource depletion and environmental impacts but also because agriculture has been recognized as key to the conservation of certain vital ecosystems and the promotion of sustainability[5]. This study focuses on the environmental sustainability of farming systems, particularly as defined and applied in existing impact assessment methods; few of these methods, however, quantitatively define environmental sustainability[6]. For example, life-cycle assessment (LCA) is a widely used indicator-based method that compiles and evaluates the inputs, outputs and potential environmental impacts of a production system. One advantage of using LCA for environmental analysis is that it considers global and local emissions linked to a service or a product[7]. It is used most often to compare production systems, identifying those that appear environmentally 'better' because they have less impact, but it does not indicate whether these systems are environmentally sustainable (e.g., if their 'polluting emissions and ... use of natural resources can be supported in the long term by the natural environment'[8]). As environmental sustainability remains such an important future-oriented concept[9], methods to define thresholds for it have been the subject of several previous studies[10–12]. However, no common definition of environmental sustainability has come to the fore in sustainability assessment.

In this paper, we propose, estimate and apply 'reference values' (RVs)[13], which explicitly define threshold levels of emissions or energy use that farming systems must not exceed to be considered environmentally sustainable.

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Unlike other methods, RVs can quantify the conceptual 'distance' to sustainability by allowing multiple targets and evaluating the influence of farm characteristics on this distance. We applied RVs to potential environmental impacts of dairy farms in Brittany, France, to assess to what extent these farms comply with the goals defined by the RVs and, more generally, to explore how RVs can be used to identify characteristics of farming systems with lower environmental impact. We also discuss potential trade-offs between environmental sustainability and food production.

## Materials and Methods

### *Impact assessment*

We used the EDEN-E tool[14], based on LCA methodology, to evaluate the environmental interventions (i.e., resource extractions, pollutant emissions and land use) and potential environmental impacts of dairy farms. According to the International Organization for Standards, LCA 'addresses the environmental aspects and potential environmental impacts (e.g., use of resources and environmental consequences of releases) throughout a product's life cycle from raw material acquisition through production, use, end-of-life treatment, recycling and final disposal (i.e., cradle-to-grave)'[15]. EDEN-E assesses the impacts of climate change, eutrophication, acidification, terrestrial ecotoxicity, land occupation and non-renewable energy (NRE) use from the acquisition of raw materials to the exit gate of the farm. Traditionally, LCA is a site- and time-independent tool that does not consider where and when environmental interventions take place, mainly because it is difficult to gather site-specific information for all sites included in an LCA[16]. Site dependence in EDEN-E was introduced by distinguishing 'direct' (on the farm site) environmental interventions from 'indirect' (associated with production and supply of inputs used on the farm) environmental interventions. The method thus distinguishes an 'on-farm' component and an 'off-farm' component for each impact considered. Potential impacts are expressed per each of three functional units: 1000 litres of milk sold, 1 'global' hectare of land occupied (on- and off-farm) and 1000 € of gross farm income. In LCA, functional units address the primary functions fulfilled by a production system[17]; for EDEN-E, these functional units represent agricultural production, land use and revenue generation, respectively. On-farm hectares correspond to a farm's usable agricultural area (UAA), whereas off-farm hectares correspond to the estimated amount of land needed to produce a farm's crop-based inputs (e.g., concentrated feed, fodder and bedding). For example, the off-farm area required to produce 1 kg of concentrated feed elsewhere than on the farm is a sum of the area necessary to produce the given quantity of each of its agricultural ingredients, which itself is a function of the ingredient's yield per hectare and post-yield processing (e.g., soybean area divided between soybean oil and soybean meal). EDEN-E contains a questionnaire to collect data about farm inputs and outputs, housing and manure-management systems, manure application techniques, crops grown, and grazing and feeding strategies. Additional data come from documents such as accounting or record books for fertilization, pest treatments and milk production. EDEN-E estimates on-farm emissions from farm-gate balances and off-farm emissions from public data[18–21] and unpublished data from the French National Institute for Agronomic Research (INRA) in Rennes. Total emissions are aggregated into impact categories using characterization methods[17].

### *Selection of environmental impacts and their indicators*

In this study, we focused exclusively on environmental impacts per hectare of land. As mentioned, one implicit function of agricultural systems is land use via the occupation of surface area. Expressing impacts per hectare rather than per unit mass of agricultural product better reflects the assumption that sustainability of agricultural systems depends on the Earth's area-limited capacity to support their emissions and use of resources. Although production efficiency is an important goal of agriculture[22], agricultural production ultimately remains more limited by the surface area available than by production capacity.

We chose to focus on three major environmental impacts of dairy farms: climate change (CC), a global impact; water-quality degradation (WQ), a regional impact; and NRE use, a resource-depletion impact. For these impacts, societal objectives have been expressed in science-based policy documents, as detailed below. CC and WQ influence important ecosystem functions, such as climate regulation, water supply and nutrient cycling[23]. Although farming systems focus on transforming solar energy into biomass, they currently depend greatly on direct and indirect inputs of NRE. Depletion of NRE sources to a point that would restrict the supply of inputs therefore could have a dramatic impact on farming systems.

CC is defined as the impact of emissions on the radiation absorption of the atmosphere[14]. Methane ($CH_4$) from enteric fermentation and manure management were calculated according to the Tier 2 method proposed by the Intergovernmental Panel on Climate Change (IPCC)[24] as a function of animal subcategories, milk production and manure-management systems. Nitrous oxide ($N_2O$) emissions were estimated by emission factors from the IPCC[25]. Gaseous N emissions from farms vary according to factors such as the total quantity of N excreted by livestock, excretion on pasture or in a building, type of manure, storage conditions and technology used in manure application on fields. Following the recommendations of other LCA studies[17], the indicator for a farm's global warming potential over a 100-year time horizon ($GWP_{100}$) was calculated as its annual greenhouse-gas (GHG) emissions expressed in kg of carbon-dioxide ($CO_2$) equivalents, based on the most recent characterization factors from the IPCC[26] in kg of $CO_2$ equivalents: $CO_2$:1, $N_2O$:298, $CH_4$:25.

**Table 1.** Mean impact values (and CV) for the 45 dairy farms and relative and normative RVs. Normative RVs for climate change: CC20 and CC50. Normative RVs for water quality: WQ50, WQ25 and WQ10. Normative RV for NRE use: NRE20.

| | Unit | Mean | Relative RV (lower quartile) | Normative RV | | |
|---|---|---|---|---|---|---|
| Climate change[1] | kg $CO_2$-eq. $ha^{-1}$ | 6107 (18%) | 5244 | CC20 | CC50 | |
| | | | | 4885 | 3053 | |
| Water quality[2] | kg N-$NO_3$ $ha^{-1}$ | 68.6 (56%) | 39.0 | WQ50 | WQ25 | WQ10 |
| | | | | 64.3 | 34.5 | 21.5 |
| NRE use[1] | GJ $ha^{-1}$ | 18 (21%) | 16.2 | NRE20 | | |
| | | | | 14.4 | – | – |

[1] Per on- and off-farm hectare.
[2] Per on-farm hectare.

NRE represents the annual quantity of NRE used by a farm, calculated in an indicator that sums in a single unit (GJ) the lower heating values of energy carriers used. All on- and off-farm processes (e.g., input manufacture and agricultural production) that used NRE sources (e.g., petroleum, coal and natural gas) were included in this calculation. For CC and NRE, both global concerns, we expressed impacts per 'global' hectare (on- and off-farm), which represents each farm's direct and indirect potential impacts.

WQ focuses on nitrate emissions, calculated as an indicator of the annual quantity of nitrogen leached as nitrate (kg N-$NO_3$) per hectare of on-farm area (excluding off-farm area, since impacts of a farm's nitrate emissions are more regional than global). We assumed that the annual quantity of nitrate-N that a farm leached into groundwater equaled its annual farm-gate N balance after subtracting gaseous N emissions (i.e., the difference between input from $NH_3$ atmospheric deposition, purchased feed and fertilizers and outputs of crops and gaseous emissions of $NH_3$, NO, $N_2O$ and $N_2$). Although this method for estimating nitrate leaching risks oversimplification and accumulation of uncertainties, previous works based on field measures[27] in the area and a Monte Carlo analysis[28] confirm its robustness.

### *RV specification*

Environmental analyses often exclude setting indicator RVs, an important step[29–31]. Among methods that include them, the Framework for Evaluating Sustainable Land Management (FESLM) method recommended setting criteria and threshold values[32]. Thresholds indicate points at which significant changes in a system may occur. More recently, authors have set RVs according to policy targets that describe a maximum critical flow[33] or to science-based objectives, both of which can demonstrate sustainability 'gaps'[34]. The main concerns, however, remain how to estimate RVs and how to use them to identify and recommend changes that could move systems closer to sustainability.

The RVs we propose fall into two categories: 'relative' and 'normative'[13]. Relative RVs are based on the average value of an indicator for a group, a region or industrial sector (at one point or over time), while normative RVs are based on target values and thresholds. Target values represent desirable conditions, while thresholds are minimum or maximum levels or ranges of acceptable values that should not be exceeded[35]. Thresholds can also be defined as values for which a minor quantitative variation would have relatively large effects on the qualitative state of the system. Relative RVs are derived from indicator values of a given population (here, a set of farms) or from a sub-population (e.g., farms having lower impacts). In contrast, normative RVs are determined either based on scientific knowledge or government policies (which are expected to have a scientific basis) or derived from values for pristine nature, a past state or the current natural environment.

We set relative and normative RVs for the three impacts considered in this study. We defined relative RVs as the lower quartile of GHG emissions (CCrel), nitrate emissions (WQrel) and non-renewable energy use (NRErel) of a sample of farms for CC, WQ and NRE impacts, respectively. Lower quartile values are familiar to farmers because extension personnel use them to define target values for technical parameters (e.g., animal growth rate and crop yield)[36]. In contrast, we based normative RVs on several science- and policy-based sources, sometimes resulting in several RVs for the same impact, as described below.

**Climate change.** For CC, two normative RVs were based on European Union (EU) commitments[37], in which France pledged to reduce its total GHG emissions by 20% by 2020 (CC20) and by 50% by 2050 (CC50). We calculated the value of these normative RVs by subtracting the percentage decrease in total or gas-specific emissions from the mean annual GHG emissions of the sample of dairy farms under consideration (Table 1).

**Water quality.** The impact of water quality on human and ecosystem health is a function of its nitrate concentration, among other characteristics[38]. In Brittany, nitrate emissions to surface water can render drinking water non-potable[39] and, once transferred to rivers, cause significant coastal eutrophication[40]. The EU's Drinking Water Abstraction Directive[41] and Nitrates Directive[42] recommend a surface-water nitrate concentration less than or

equal to 25 mg $NO_3 l^{-1}$ and mandate a maximum concentration of 50 mg $NO_3 l^{-1}$, which we adopted as two normative RVs (WQ25 and WQ50, respectively). We based a third normative RV, focused on ecosystem health, on research by Ménesguen[40] in Lannion Bay, one of the sites most affected by coastal algal blooms in Brittany, where greatly reducing algal blooms would require a maximum concentration of 10 mg $NO_3 l^{-1}$ in surface waters (WQ10).

To estimate the per-hectare amounts of farm nitrate leaching that would lead to reaching these three nitrate-concentration limits, we reversed a method, proposed by Payraudeau et al.[27], that estimated the mean annual nitrate concentration of water draining from agricultural fields in Brittany as a function of annual effective rainfall (precipitation minus evapotranspiration) and farm nitrate. Transforming these limits of nitrate concentration in regional surface waters into per-hectare RVs of farm nitrate losses first required considering the influence of wetlands and non-agricultural land uses. Since we were interested in coastal eutrophication, denitrification was considered at a scale larger than the farm. At watershed scales and above, wetland denitrification can decrease nitrate concentrations by approximately 1 mg $l^{-1}$ per percent of wetland surface present[43] (P. Merot, personal communication). In Brittany, wetlands represent 10% of the surface area. Next, we assumed that 20% of Brittany's surface area does not leach nitrates (e.g., forests and fallow agricultural land); thus, water flowing from this area would dilute surface-water nitrate concentrations (P. Durand, personal communication). Consequently, the three nitrate concentration limits were increased by 10 mg $l^{-1}$ to account for eventual wetland denitrification, then divided by 0.8 to estimate nitrate limits only for the land from which nitrate leaches. To represent long-term temporal variability, mean annual effective rainfall over the past decade in Brittany was used as input data[44]. We did not consider denitrification at the farm scale and assumed that the mineral N content of the soil remained constant over the long term. Based on these assumptions, nitrate concentrations of 50, 25 and 10 mg $NO_3 l^{-1}$ for normative RVs corresponded to 64.3, 37.5 and 21.5 kg of nitrate-N leached per on-farm hectare, respectively (Table 1).

**NRE use.** The EU proposed a target of 20% of gross inland energy consumption coming from renewable sources by 2020[45] to reduce GHG emissions and slow climate change. In consequence, we estimated a normative RV (NRE20) equal to 80% of the mean annual NRE use of the sample of dairy farms under consideration (Table 1).

### *Application to dairy farms*

We calculated RVs for this study based on the estimated emissions and NRE use of 45 dairy farms in Brittany. The 39 conventional and six organic farms belong to the ETRE and BIO dairy-farm networks in Brittany, respectively, which consist of farms selected for above-average economic performance. These networks aim to share the data necessary to analyze and characterize the functioning and management of these farms and draw lessons for economically challenged dairy farms (F. Roger, personal communication). These farms represent a subset of the 47 farms analyzed in van der Werf et al.[14] for which sufficient data were available regarding crop rotations, land use for animal and crop production, and quantities of crops sold. In Brittany, most dairy farms produce both crops and animal products, except for dairy farms based solely on pastures. Management of the milk-production system thus depends on crop production, which either can be fed to cattle on the farm or sold. We decided not to exclude non-dairy production because the dairy farms studied relied on interactions between animal and crop production. Thus, to aggregate milk with other products that could be consumed by humans, we calculated the nutritional energy and protein contents of the plant and animal products sold[46,47] and summed each set to estimate total production of energy and protein. Both were expressed per ha of on- and off-farm area occupied because milk production required crop input from off-farm fields.

For each impact category and RV, we placed farms into two groups: those whose potential impact per ha exceeded the RV ('above') and those for which it did not ('below'). Results of this step led us to continue the analysis with only the most severe RV for each impact category for which at least six farms qualified: CC20, WQ25 and NRE20. We calculated and compared mean characteristics, inputs, outputs and potential impacts for each group, identifying significant ($P < 0.05$) differences with the non-parametric Mann–Whitney test.

## Results

Variability among farms was large for the impacts considered and for nutritional energy production, with coefficients of variation ranging from 18% for CC to 56% for WQ (Table 1, Fig. 1). GHG emissions per hectare (on- and off-farm) varied from 3979 to 8380 kg $CO_2$ eq., N-$NO_3$ emissions per on-farm hectare varied from 0 to 165 kg, NRE per hectare (on- and off-farm) varied from 6.7 to 25.6 GJ and nutritional energy production per hectare (on- and off-farm) varied from 8.3 to 65.5 GJ (Fig. 1). Organic farms had lower mean impacts and nutritional energy production than conventional farms (Fig. 1). Values for the three impacts were significantly but weakly correlated with each other ($r^2 = 0.13$–$0.19$), while total protein and nutritional energy production each were significantly, but weakly, correlated with NRE ($r^2 = 0.14$ and 0.18, respectively). The strongest correlation observed ($r^2 = 0.49$) was that between NRE and total protein production.

By definition (i.e., lower quartile of the population), 11 farms were below each of the relative RVs (Fig. 1). For normative RVs, seven farms were below CC20, whereas none was below CC50. For WQ, 22 farms were below WQ50, 11 below WQ25 and three below WQ10. For NRE,

**Table 2.** Number of farms in a sample of 45 below relative and normative RVs for CC, WQ and NRE use impacts.

| | Climate change[1] | | | Water quality[2] | | | | NRE use[1] | |
|---|---|---|---|---|---|---|---|---|---|
| **Farms** | **CCrel** | **CC20** | **CC50** | **WQrel** | **WQ50** | **WQ25** | **WQ10** | **NRErel** | **NRE20** |
| Organic | 4 | 4 | – | 4 | 4 | 4 | 3 | 6 | 3 |
| Conventional | 7 | 3 | – | 7 | 18 | 7 | – | 5 | 3 |
| Total | 11 | 7 | – | 11 | 22 | 11 | 3 | 11 | 6 |

[1] Per on- and off-farm hectare.
[2] Per on-farm hectare.

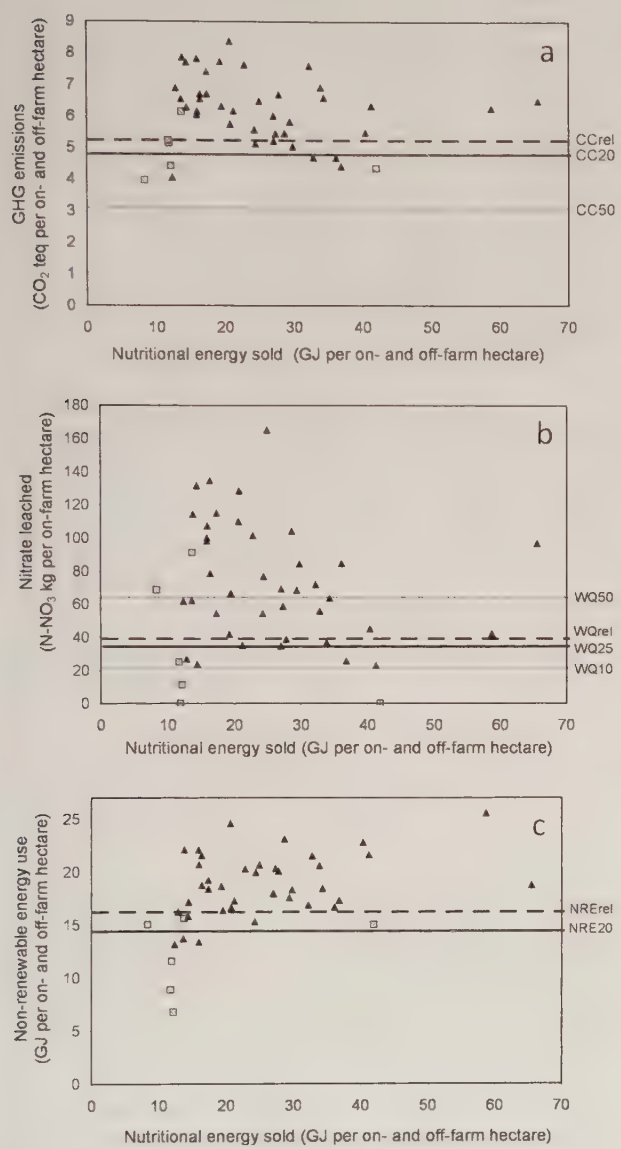


**Figure 1.** Relative (dashed lines) and normative (solid lines) RVs superimposed on estimates from the 45-farm sample of organic (squares) and conventional (triangles) dairy farms for (a) GHG emissions, (b) nitrates leached and (c) NRE use, all expressed by nutritional energy sold.

six farms were below NRE20 (Table 2). Compared to farm groups above the normative RVs CC20, WQ25 and NRE20, groups below each of them had a higher percentage of organic farms, larger mean UAAs, longer mean pasture residence time and usually a lower input level, especially of concentrate feed and nitrogen (Table 3). The mean percentage of fodder crops and grass area (FCGA) in the UAA was significantly larger for the group below NRE20, whereas the percentage of silage maize area in the FCGA was significantly smaller for this group (Table 3). The contribution of animal products to total nutritional energy produced per hectare varied from 19 to 100%, with total nutritional energy decreasing exponentially as the fraction provided by animal products increased (Fig. 2).

A farm group below a given RV tended to have lower mean milk production per cow and per hectare; consequently, it produced significantly less mean nutritional energy per hectare from animal products than the group above the same RV, but only the group below CC20 produced significantly less protein per hectare from animal products (Table 4). As for mean total nutritional energy and protein, only the group below NRE20 produced significantly less than the group above it, due to its near-zero mean production of plant products (Table 4). Farms below WQ25 and CC20 were found across the range of animal-product share in total nutritional energy produced, whereas, as mentioned, farms below NRE20 produced almost exclusively animal products (Fig. 2).

Nearly all groups below a given RV had significantly lower mean $CO_2$, $N_2O$ and total GHG emissions than the group above the corresponding RV (Table 5). The difference in mean total GHG emissions between groups below and above a given RV was largest for CC20 (2039 kg $CO_2$-eq. $ha^{-1}$), mainly due to the difference in mean $CH_4$ emission. For WQ25, lower (but non-significant) GHG emissions resulted mainly from significantly lower mean emission of $N_2O$, whereas for NRE20 they resulted from significantly lower mean emissions of $CO_2$ and $N_2O$. The difference in nitrate leached between groups below and above a given RV was larger for WQ25 (61.6 kg $ha^{-1}$) than for CC20 (29.2 kg $ha^{-1}$) or NRE20 (29 kg $ha^{-1}$). The difference in NRE between groups below and above a given RV was larger for NRE20 (7.7 GJ $ha^{-1}$) than for CC20 (3.5 GJ $ha^{-1}$) or WQ25 (3.3 GJ $ha^{-1}$) (Table 5).

## Discussion

### *Correlations among impacts and production*

The significant correlation between each pair of impacts indicates the existence of a positive relation between them

**Table 3.** Annual mean values of characteristics of the 45 farms (All) and subgroups whose impacts were below or above RVs for CC20, WQ25 and NRE20 use. Differences between farms below or above RVs are significant at $P < 0.05$ (*).

| | | | CC20 | | WQ25 | | NRE20 | |
|---|---|---|---|---|---|---|---|---|
| **Characteristic** | **Unit** | **All** | **Below** | **Above** | **Below** | **Above** | **Below** | **Above** |
| Number of farms | | 45 | 7 | 38 | 11 | 34 | 6 | 39 |
| Percentage organic farms | % | 13 | 57* | 8 | 36* | 6 | 50* | 8 |
| UAA | ha | 60 | 74* | 57 | 68* | 57 | 69* | 59 |
| Off-farm area | ha | 8 | 7 | 8 | 4* | 10 | 6 | 8 |
| On and off-farm areas | ha | 68 | 81* | 65 | 72 | 67 | 75* | 67 |
| FCGA | ha | 46 | 55 | 44 | 49 | 45 | 63* | 43 |
| FCGA in UAA | % | 77 | 74 | 77 | 72 | 80 | 91* | 73 |
| Silage maize area in FCGA | % | 26 | 20 | 28 | 21 | 28 | 10* | 29 |
| Stocking density in FCGA | LU[1] $ha^{-1}$ | 1.45 | 1.34 | 1.47 | 1.48 | 1.44 | 1.21* | 1.49 |
| Pasture residence time[2] | days | 205 | 225 | 202 | 215 | 202 | 233* | 201 |
| Concentrate feed use per cow | kg | 691 | 533 | 720 | 461* | 765 | 429* | 731 |
| Total N input per UAA | kg $ha^{-1}$ | 143 | 107* | 149 | 90* | 160 | 92* | 151 |
| N balance surplus | kg $ha^{-1}$ | 83 | 48* | 89 | 33* | 99 | 53* | 87 |
| Diesel use per UAA | kg $ha^{-1}$ | 100 | 79 | 104 | 106 | 98 | 65* | 106 |
| Electricity use per UAA | kWh $ha^{-1}$ | 333 | 342 | 331 | 221* | 370 | 235* | 348 |

[1] LU: French livestock unit[48].
[2] Pasture residence time per day with at least 8 h of pasture, summed and converted to days.
FCGA, fodder crops and grass area; UAA, usable agricultural area.

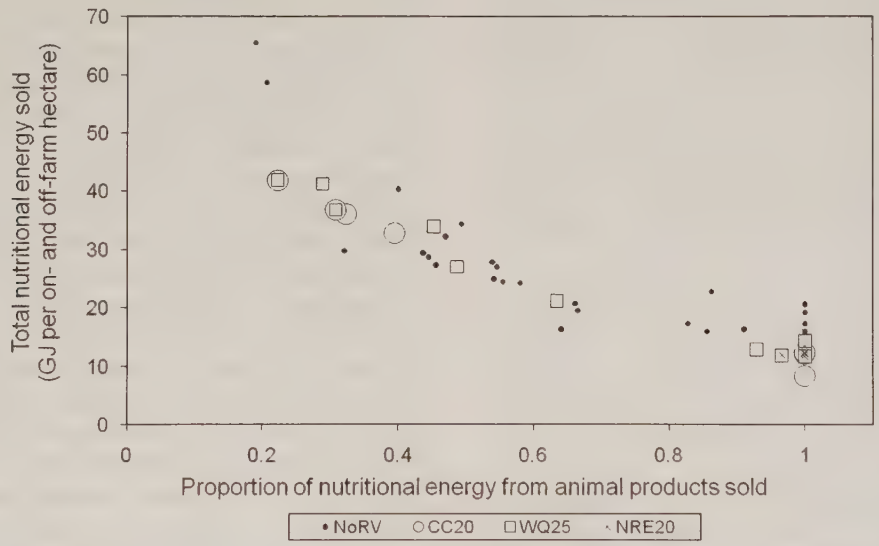


**Figure 2.** Distribution of farms according to the proportion of total nutritional energy sold that came from animal products (milk and meat). Circles, squares and crosses represent farms below the CC, WQ and NRE use RVs, respectively, while black dots represent farms above all three RVs. Certain farms are represented by two symbols.

(i.e., lower impacts in one category did not necessarily come at the cost of higher impacts in another category), but their low strength reflects the high inter-farm variability. In contrast, the relatively strong correlation between NRE and total protein production indicates the high dependence of food production on external fossil-energy inputs (e.g., diesel, machinery and fertilizers).

## *Differences between farm groups*

Groups of farms below RVs used fewer inputs per unit area, resulting in less undesired output (e.g., GHGs and nitrate); however, the group below NRE20 also produced significantly less food per unit area. These farms (half of them organic), with the longest pasture residence time, the lowest stocking density, the lowest concentrate feed use per cow and the lowest diesel use per ha of UAA, focused fully on low-input grazing-based milk production, thereby producing significantly less nutritional energy and protein per hectare due to the lack of crop production.

Attaining the WQ25 RV was a feasible objective for organic (4/11) and conventional farms (7/11), since a small surplus N balance, important for ensuring sufficient N availability for crops, was conserved. These farms have the

**Table 4.** Annual mean production values per hectare of on- and off-farm areas for all farms (All) and subgroups whose impacts were below or above RVs for CC20, WQ25 and NRE20 use. Differences between farms below or above RVs are significant at $P < 0.05$ (*).

| | | | CC20 | | WQ25 | | NRE20 | |
|---|---|---|---|---|---|---|---|---|
| **Products** | **Unit** | **All** | **Below** | **Above** | **Below** | **Above** | **Below** | **Above** |
| Milk production[1] | kg cow$^{-1}$ | 7391 | 6814 | 7498 | 6515* | 7675 | 6628* | 7509 |
| Milk production[1] | kg ha$^{-1}$ | 4578 | 3658* | 4748 | 4091* | 4736 | 4224 | 4633 |
| Total nutritional energy (NE) | GJ ha$^{-1}$ | 24.5 | 25.8 | 24.2 | 24.1 | 24.6 | 12.9* | 26.3 |
| Animal-product NE | GJ ha$^{-1}$ | 13.7 | 11.1* | 14.2 | 12.4* | 14.1 | 12.8* | 13.8 |
| Plant-product NE | GJ ha$^{-1}$ | 10.8 | 14.6 | 10.1 | 11.7 | 10.5 | 0.1* | 12.4 |
| Animal products in total NE | % | 68 | 61 | 69 | 66 | 69 | 99* | 63 |
| Total protein | kg ha$^{-1}$ | 253 | 259 | 252 | 249 | 254 | 149* | 269 |
| Animal-product protein | kg ha$^{-1}$ | 158 | 130* | 163 | 144 | 163 | 148 | 160 |
| Plant-product protein | kg ha$^{-1}$ | 107 | 143* | 101 | 114* | 105 | 1* | 124 |
| Animal products in total protein | % | 72 | 65 | 73 | 70 | 73 | 99* | 68 |

[1] Fat- and protein-corrected milk, i.e., $0.337 + 0.116 \times \text{fat} + 0.06 \times \text{protein} \times \text{kg milk produced}$[49].

**Table 5.** Estimated annual mean GHG emissions, nitrate leached and energy use per hectare for the 45 farms (All) and subgroups whose impacts were below or above RVs for climate change (CC20), water quality (WQ25) and non-renewable energy use (NRE20) use. Differences between farms below or above RVs are significant at $P < 0.05$ (*).

| | | | CC20 | | WQ25 | | NRE20 | |
|---|---|---|---|---|---|---|---|---|
| **Emission/impact** | **Unit** | **All** | **Below** | **Above** | **Below** | **Above** | **Below** | **Above** |
| $CO_2$ emission[1] | kg $CO_2$ ha$^{-1}$ | 899 | 723* | 932 | 782* | 937 | 538* | 955 |
| $N_2O$ emission[1] | kg $CO_2$-eq. ha$^{-1}$ | 1842 | 1335* | 1935 | 1528* | 1943 | 1448* | 1902 |
| $CH_4$ emission[1] | kg $CO_2$-eq. ha$^{-1}$ | 3366 | 2327* | 3557 | 3352 | 3370 | 3268 | 3381 |
| Climate change[1] | kg $CO_2$-eq. ha$^{-1}$ | 6107 | 4385* | 6424 | 5662 | 6250 | 5254* | 6238 |
| Nitrate leached[2] | kg N-$NO_3$ ha$^{-1}$ | 68.6 | 43.9* | 73.1 | 22.1* | 83.7 | 43.5* | 72.5 |
| NRE use[1] | GJ ha$^{-1}$ | 18.0 | 15.1* | 18.6 | 15.5* | 18.8 | 11.3 | 19.0 |

[1] Per on- and off-farm hectare.
[2] Per on-farm hectare.

lowest off-farm area used, and a relatively low proportion (70%) of protein produced from animal products. In French dairy systems, some reduction in WQ impacts can be reached without major changes by optimizing feed strategies and pasture management or reducing stocking rates[50].

Farms below the CC20 RV produced significantly less nutritional energy and protein from animal products and more from plant products than the farms above this RV. The CC20 RV was attained mainly not only by organic systems (4/7) but also by some highly intensive conventional systems. For the organic systems, lower emissions per hectare were explained by their larger size and relatively low use of inputs. For intensive conventional systems, it was closely linked to increased milk production per cow, which can reduce GHG emissions by producing the same quantity of milk with fewer cows[51].

Variability among farms within each production mode makes analysis of differences between organic and conventional farms difficult. Results seem to indicate, however, that all systems may move toward environmental sustainability even though, according to production mode and intensity, some potential impacts are easer to reduce than others. For instance, more conventional farms attained the WQ RV than attained the CC RV, while the conventional farms attaining NRE RV were exclusively grass-based systems.

### *Defining RVs*

Fortunately, legislative norms and policy objectives were available to help define normative RVs. The difference in the dates of origin of these objectives reflects longstanding concern over WQ and more recent worries about CC. For simplicity, we assumed a reduction target of the normative RVs for dairy farms equal to the overall reduction target (e.g., 20% fewer GHG emissions by 2020). One can, however, define RVs based on sector-specific targets, which may require, for example, the transportation sector of an economy to reduce emissions or energy use more than the agricultural sector. Interestingly, the two strictest WQ RVs (WQ25 and WQ10) represented relative reductions (56 and 69%, respectively) similar to that for the strictest CC RV, CC50 (50%). Despite this relative similarity, no farms attained CC50, whereas three farms (all organic) attained WQ10; the lower variability of CC than WQ impacts reflects the greater difficulty in reducing GHG emissions on

farms than nitrogen losses. Although the relatively modest reduction (20%) of the NRE normative RV (NRE20) placed it within reach for six farms, farms that attained it produced hardly any plant products, unlike those that attained other RVs.

The fact that dairy farms in the sample had above-average economic performance (implying better farm management) suggests that our relative RVs may be lower than those that would be calculated for a larger, more representative sample of French dairy farms. In addition, the ratio of organic to conventional farms in the sample of 6 : 39 (15%) is larger than that found among all dairy farms in Brittany (2%), which may have influenced the values of relative and certain normative RVs. Also, our normative RVs for CC and NRE were not independent of the values in the farm sample. Thus, given the influence of sample means on RVs, it appears more appropriate to calculate them using larger and more representative sample sizes, such as region- or country-level means of a given type of farm.

More accurate RVs may be established by fate modeling, especially for regional impacts such as water quality, where landscape plays an important role. For instance, WQ was complicated to estimate due to its local nature. Although using farm-gate N balance (including deposition and volatilization) to estimate nitrate pollution may be an interesting means-based indicator of the N-management efficiency of farms[52], it does not consider the influence of environmental factors such as soil and climate on pollutant fate. We suggested a simple method to estimate denitrification and dilution processes at the watershed scale to set the WQ RV, converting a stream nitrate concentration into an equivalent amount of nitrogen leached from fields.

Although systems should be evaluated periodically to chart their progression toward sustainability[53], RVs must be established for the long term. Thus, whenever possible, annual variability in potential impacts should be taken into account when defining RVs. In the case of WQ, according to previous studies in Brittany[54], inter-annual variation in nitrate leaching can be explained mainly by two factors: soil-N stocks at the beginning of winter and annual mean rainfall. The first depends on crops, crop management and soil, which were assumed in our study not to vary over the long term. The second was included by using the mean annual effective rainfall during the past decade in the region[44]. To improve estimation of this RV, these factors could be represented better with simulation models such as Melodie[54], a whole-farm model, or TNT2[55], which simulates nitrogen dynamics in watersheds.

All hypotheses and assumptions made to define an RV must be explained clearly to refine RVs in the future. Developing RVs based on ecological thresholds could be an interesting future goal[10]. Priority should be given to normative RVs when enough data exist to set them. Multiple RVs are consistent not only with multiple sources of knowledge but also with graduated stages toward sustainability goals. Fricker[2] and Fischer et al.[56] introduced a temporal scale to attain sustainability that varies over time. Thus, while we have objectives to attain in the next 20 years, we should keep in mind objectives needed to maintain sustainability in 50 or 100 years. Reaching long-term objectives may require more drastic changes in farming systems and society than reaching short- or medium-term objectives.

Other, more ambitious, goals could be used to define normative RVs. An earlier French objective, suggested by a workshop of experts from several fields (e.g., academia, industry and NGOs), aimed to reduce GHG emissions by 75% by 2050[57]. This goal, however, does not consider what proportion of the overall reduction should come from reducing emissions of each of the three main GHGs ($CH_4$, $N_2O$ and $CO_2$). In contrast, Anderson and Bows[58] argue that, specifically, $CO_2$ emissions from fossil-energy use and deforestation should be eliminated to avoid dangerous CC (i.e., increasing mean global temperature >2°C). Since emissions of the other two GHGs ($CH_4$ and $N_2O$) are associated with agriculture (and therefore impossible to eliminate), they proposed reducing them by 50% by 2050[58]. Dairy systems evaluated in our study, however, could reduce, but not halve, their $CH_4$ emissions if they want to maintain the same milk-production level.

### *Applying RVs*

Relative RVs may be easier to implement in policies because farmers may prefer being evaluated in relation to their current performance and that of their peers. They also may prefer, however, to be evaluated according to RVs for their mode of production (e.g., organic). For example, conventional farmers may balk at RVs that were calculated with emission values per hectare from organic farms. Similarly, pig farmers may not wish to follow the same RVs as dairy farmers, and vice versa. Complicating things further, the number and type of farms in a given area (e.g., watershed) may influence the RV that each of them has to follow (e.g., areas with more forest and areas with more pig farmers). Plus, when farm-site characteristics such as groundwater depth or climate influence local impacts, farmers may call for methods of RV calculation that explicitly consider these site characteristics.

### *Sustainable agricultural systems and food production*

There is no single path to sustainability of any farm type. Analysis of technical and management changes is an important step toward sustainable systems[31]. The characteristics of low-impact dairy farms identified in this study (low energy and N input for organic systems, high milk production per cow for intensive conventional systems) could be adopted by other dairy farms to reduce the potential CC and WQ impacts of milk production without necessarily decreasing the total amount of nutritional energy or protein produced. Other management options for decreasing CC and WQ impacts exist, however, such as producing more total nutritional energy by decreasing

milk production and increasing crop production for human consumption. As our results indicated, however, reducing NRE while maintaining milk- and especially crop-production levels is more difficult due to the relatively strong correlation between farm inputs and productivity.

Expressing farm production by total nutritional energy considers all farm products and responds to the desire to connect agricultural research with nutritional sciences[59–62], especially now that CC threatens the ability to feed an increasing population with limited natural resources. According to our results, management options that would decrease milk production in favor of crop production would not require a major transformation of systems for dairy farms that already produce and sell crops. Alternative production scenarios may be more limited for pasture-based dairy farms that have no crop production. These farms may explore other options, such as optimizing pasture management and continuing to reduce inputs.

Currently, reconciling human food security and environmental sustainability may depend on the ability to adapt farming systems to produce the same amount of nutritional energy with a larger proportion of crops and less animal production. In environmental assessment, quantifying nutritional energy and having the surface area used (on- and off-farm) as the unit of environmental assessment help identify low-impact farms without focusing on a particular agricultural product. This is especially important for farming systems that produce several products (e.g., milk, meat and crops).

Per hectare RVs can supply complementary information to stakeholders involved in environmental assessment of agricultural products. Currently, many countries are in the process of placing carbon labels on their agricultural products. As carbon labels express CC impacts per unit of mass, potentially favoring highly productive systems, the impact-per-area approach of defining low-impact systems that RVs provide may serve as an appropriate complement.

## Conclusion

RVs are an attempt to measure environmental sustainability by accounting for environmental requirements and selecting systems that respect those requirements, in contrast to current evaluation methods, which merely identify systems that pollute less. RVs are a useful approach to quantify and represent how far systems may lie from sustainability policies, regulations or goals. They can be used in an iterative process to guide farming systems to environmental sustainability. RVs can be used at the farm scale as demonstrated in this paper; they also can be used at a regional scale to explore scenarios of sustainable development. Although RVs may be difficult to establish, they render sustainability goals more concrete and operational.

The use of RVs has several advantages: (i) clearly expressing which sustainability objectives have been chosen; (ii) quantifying the conceptual 'distance' between current systems and those objectives; (iii) identifying technical and management practices to improve in existing systems; and (iv) working with common existing indicators. Once RVs have been defined, they can be adapted to a variety of environmental assessment methods since they are established for individual indicators.

Reconciling human food security and environmental sustainability may depend on the ability to adapt farming systems to produce the same amount of nutritional energy with a larger proportion of crop products and fewer animal products. The distance between normative RVs and indicator values of dairy farms suggest that production and consumption of agricultural products will need to change if sustainability goals do not.

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# Technical efficiency of organic pasture farming in Germany: The role of location economics and of specific knowledge

S. Lakner*, S. von Cramon-Taubadel and B. Brümmer

Department of Agricultural Economics and Rural Development, Georg-August University Göttingen, Platz der Göttinger Sieben 5, D-37073 Göttingen, Germany.
*Corresponding author: slakner@gwdg.de

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## Abstract

We analyze the efficiency of organic pasture farming in Germany using data from 1994/95 to 2005/06. Five inputs and one output are analyzed by means of a stochastic frontier production function, allowing for heteroscedasticity and technical effects. Five sets of possible determinants of technical efficiency are considered in the model. These include: (1) farm structure and resources; (2) human capital and management capacities; (3) institutional choice; and (4) subsidies. To these factors that are commonly included in technical effects models, we add (5) a set of variables that capture localization and urbanization economies such as the share of organic farms in a region and the regional share of votes for the Green Party in recent elections. These regional effects are found to have a significant impact on the technical efficiency of organic farms. The evolution of efficiency on farms that are converting from conventional to organic farming is also analyzed.

**Key words:** efficiency analysis, organic farming, location economics, specific knowledge

## Introduction

The importance of organic farming has increased rapidly in the EU and especially Germany in recent years. The share of organic farmers in Germany grew from 1.3% in 1996 to 5.9% in 2010[1]. Currently, 7.9 million hectares or 4.5% of the agricultural area in the EU is under organic production[2]. As concerns rise about agriculture's contribution to climate change, its influence on biodiversity, and issues such as animal welfare and food safety, EU and member state policy-makers can be expected to direct more attention and probably support toward organic farming.

Designing appropriate policies requires a solid understanding of the factors, such as technical efficiency (TE) that influence the competitiveness of organic farming. Due to the regulations that define and govern organic farming, organic production systems differ from conventional systems, and are subject to different relative scarcities. For example, since synthetic fertilizer use is not permitted in organic farming, nitrogen scarcity forces organic farmers to make efficient use of organic manure or nitrogen fixed, for example, by clover or other leguminous plants[3]. The greater dependence of organic production systems on the availability of natural resources could lead to a higher variance of TE scores in organic farming[4].

The organic production regulation promotes more integrated farming systems. A study on organic dairy farms in the USA shows that on organic dairy farms the vertical integration produces substantial economies of scope (in contrast to conventional dairy farms). There seems to be a systematic incentive toward an integrated farming and against a high degree of specialization[5], which is a special characteristic of organic farming systems too.

These issues warrant a closer look at efficiency in organic farming, and at the possible determinants of differences in efficiency between organic farms. Besides the standard determinants that are usually considered in efficiency analysis—such as farm structure or the age and educational status of the farm manager—we highlight two factors that have received little attention in the literature on the efficiency of organic farming to date.

The first of these factors is a *farm's conversion status*. Nieberg[6] shows that grain yields fall to 53% of their starting level after 4 years on German farms that are converting from conventional to organic farming, and then recover slightly to 72% of the starting level by the sixth year. Farmers in conversion have to acquire specific

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knowledge and expertise for managing a new technology. Hence, one might expect TE to fall on a farm that has begun to convert, but to increase thereafter as the farmer acquires knowledge and expertise. This has implications for policy, if increasing the number of organic farms or the share of organic food production is a policy goal and policy-makers wish to design appropriate support packages for converting farms.

The second factor that has received little attention is *location*. In the past century, organic farming in Germany emerged in several distinct regional centers. The 'Demeter' movement influenced by Rudolf Steiner (1861–1925) started in the 1920s on east German farms. In the 1960s and 1970s, many small farms in southern Germany and Switzerland followed the ideas of Hans Müller (1891–1988) and Hans-Peter Rusch (1906–1977). Finally, after German reunification, many farms in the former German Democratic Republic converted to organic farming with the support of EU agri-environmental schemes that were introduced with the 'MacSharry Reform' of 1992[7]. The emergence of organic farming in distinct geographic regions suggests that location effects such as localization and urbanization economies effects might have played a role in the development of the sector[8] via their impact on farm-level efficiency.

In this paper, we analyze the determinants of the efficiency of organic pasture farms in Germany. Extending existing studies, we address two important questions for organic farming in Germany and elsewhere; whether location has a significant impact on TE and whether TE follows a distinct path over the course of the conversion period from conventional to organic farming. We proceed as follows: in the 'Background/Literature Survey' section, we provide a short overview of the recent literature on the impact of conversion and location effects on TE, and we propose a framework that includes location effects among the determinants of TE on organic pasture farms. In the 'Methods and Data' section, we present the methods and the data used in our empirical analysis. In the 'Results and discussion' section, we present and discuss the results, and the 'Conclusion' section concludes.

## Background/Literature Survey

An extensive literature deals with the determinants of TE in farming in general[4,9–14]. This literature has identified the following four categories of determinants of TE:

1. farm structure and available natural resources;
2. management capacities and human capital;
3. institutional choice (legal form and tax options); and
4. market orientation and policy support.

Several studies investigate the TE of organic farms specifically[4,15–20]. The development of TE during the conversion period from conventional to organic farming is addressed in some of these studies. Lohr and Park[19] studied the influence of the organic farm manager's experience in the USA by splitting the sample into farms with more than 5 years of experience in organic farming, and farms with less than 5 years. The authors found that efficiency grows with years of experience in organic production. According to Sipiläinen and Oude Lansink[17], the learning process in organic dairy farms in conversion in Finland takes about 6–7 years following the decision to convert.

Initial studies of the location of organic farming in the 1980s and 1990s hypothesized that organic farms would cluster in the neighborhoods of cities with high population densities and therefore a high potential for direct marketing[21]. After the introduction of per-hectare payments for organic farming by EU regulation 2078/92 in 1993, many farms converted in parts of east Germany where agro-climatic conditions are less favorable[21]. One determinant of the spatial distribution of organic farming might therefore be lower opportunity costs due to less profitable conditions for conventional farming.

Bichler et al.[8] confirmed that there is a higher concentration of organic farms in German regions with less favorable soil quality. They also found a higher prevalence of organic farming in regions in which the difference between the subsidies provided for organic and for conventional pasture is large, and in regions in which greater shares of land have been designated as water protection or nature conservation areas. In some west German regions, local per capita income and farm size were also found to have a positive influence on the share of organic farms. Finally, Bichler et al.[8] also provide evidence that the spatial distribution of organic farms tends to be more concentrated in proximity to big cities with high population densities. In Figure 1, we can see the local share of organic farms in the total number of farms at the county level in 2007, which in general shows the local concentration of organic farming, which is in the south, center-west and north-east of Germany.

A number of studies have analyzed the role of information spillovers on the spatial distribution of organic farming. Farmers in the UK who are converting from conventional to organic production receive most of their information from other organic farmers in the same region[24]. Studies in Germany[7] and Wales[25,26] confirm that neighboring organic farmers are the main source of information for converting and converted organic farms. Regional networks of farmers and support groups encourage the regional concentration of organic farms in Wales and England[27]. The importance of information exchange between farmers is probably greater for organic than for conventional farming, because until recently there were relatively few specialized extension and education/training services for organic farmers. This is generally true for the western European countries that we have studied, and certainly for Germany. Therefore, most knowledge and information had to be generated by organic farmers themselves and was disseminated via local working groups[7]. Where specialized extension services exist, they were found to have a positive impact on organic conversion rates[28].

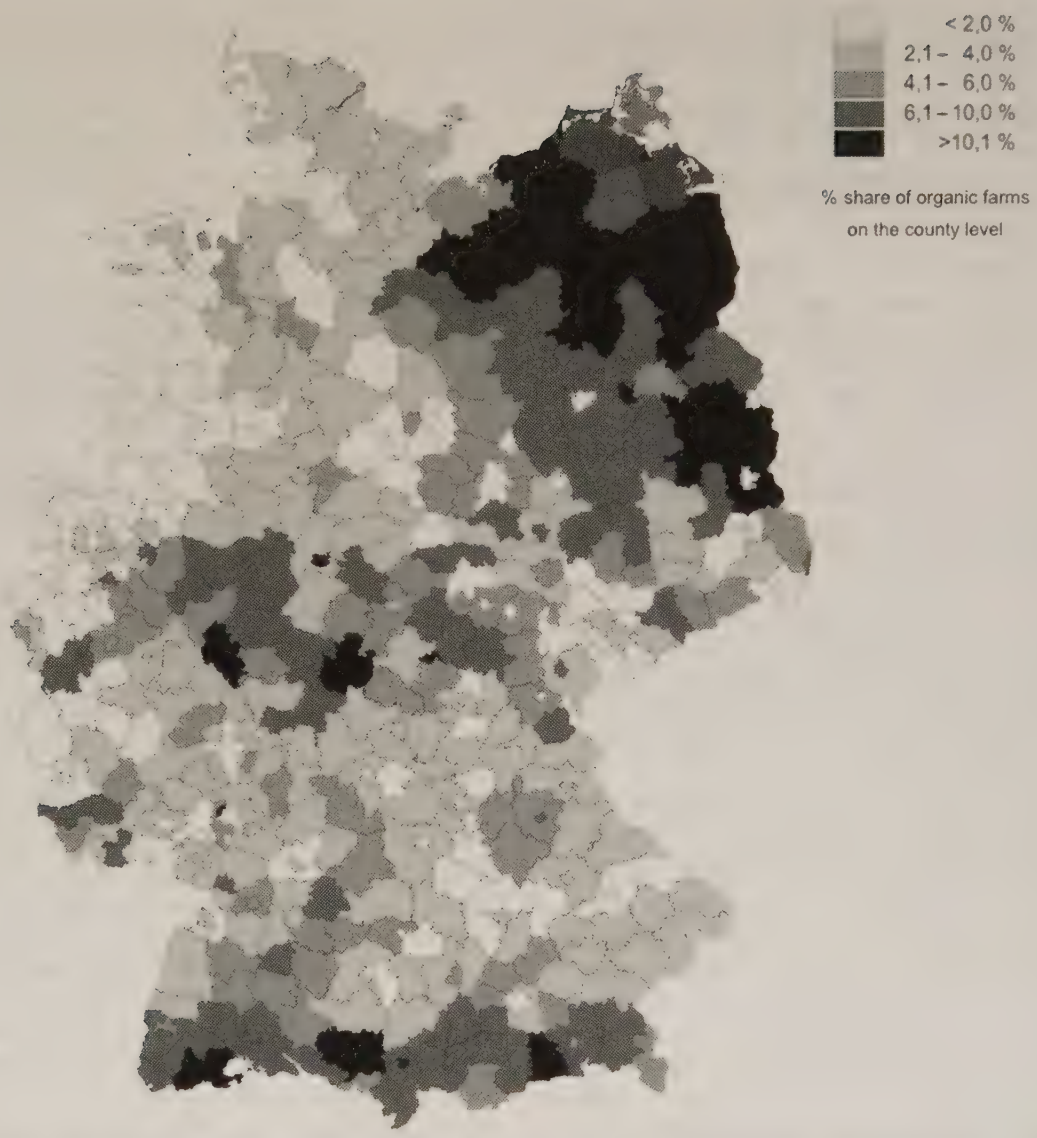


**Figure 1.** Regional concentration of organic farms in Germany in 2007. Source: Own calculation. Data on the county ('Landkreis') level from the German Federal Statistical Office[22], Regional Database 2005 and 2007[23]. Due to administration reforms, in 15 of 457 counties there were no data in the regional statistics for the year 2007. For the missing data in 2007, we took data from the previous years 2005 or 2003, assuming that the organic sector is constantly growing and we get a 'minimum number' of organic farms from 2003/2005 for the year 2007.

If information spillovers between organic farms are important, it is reasonable to expect that location will have an impact on farm-level efficiency. While we are aware of no studies on the impact of location on the efficiency of organic farms, the impact of location on efficiency has been studied in other agricultural contexts. For example, according to one study, a high regional concentration of aquaculture farms in Norway is associated with lower output but higher efficiency[29]. In other studies, the total sales per employee increase with the number of organic processors and traders located in a county in the USA[30], and regional concentration, distance to the nearest slaughterhouse, and the population density in neighboring regions influence the TE of pig farms in western France[31]. There is also evidence of spatial-efficiency clusters in Ukrainian dairy farming, with proximity to other efficient farms and to dairy processors exercising a positive effect on efficiency and total factor productivity[32].

Based on this review of the literature, we propose to consider—in addition to the four 'traditional' categories of factors that influence efficiency mentioned above—variables that account for location effects. First, a large share of organic farmers in a region might create external advantages that are referred to as *localization economies* in location theory[33]. Localization economies arise when many firms from the same industry are located in a region. They include access to pooled markets for skilled labor, the regional emergence of firms that provide specific intermediate inputs, and ease of communication leading to technological spillovers. All of these factors are expected to

have a positive impact on TE as the number of organic farmers in a region increases.

Second, the efficiency of organic farming might also benefit from *urbanization economies*[34]. Urbanization economies are the effects on a firm's economic performance that arise from regional economic activity in other sectors. At first glance, it may seem incongruous to discuss urbanization in connection with a distinctly rural activity such as organic farming. However, in a densely populated and comparatively decentralized country such as Germany, rural and urban areas lie close together and are strongly interlinked. It is reasonable to expect that organic farms will perform better in regions that are characterized by local communities that are sensitive to environmental concerns and willing to pay higher prices for organic products. Such regions will provide sufficient demand to sustain direct marketing, specialized retail outlets such as organic farmers' markets and restaurants that specialize in organic food, and processors such as organic bakers and dairy enterprises that benefit from a dependable local supply of organic farm products. Such regions will also be able to sustain special advisory services for organic farms, and service providers that cater to the needs and preferences of organic farmers (e.g., a Waldorf school for children, practitioners of alternative medicine). We capture urbanization effects with two variables. Since we study organic pasture farms, we use a farm's distance from the nearest organic dairy processor as an indicator of local-demand conditions. In addition, we use recent local election results of the Green Party as a measure of the overall local level of acceptance for organic farming. The Green Party in Germany has traditionally championed environmental issues and organic farming, and most organic farmers and their supporters come from a socio-economic background that is associated with support for the Green Party[7].

## Methods and Data

### Methods

The field of empirical efficiency analysis in economics has grown very rapidly over the past roughly two decades, with applications not only in agriculture but also in many other areas such as health care provision and energy generation. Two main empirical approaches have been developed; so-called Data Envelopment Analysis (DEA) based on programming techniques and Stochastic Frontier Analysis (SFA) based on econometric estimation of production functions. We employ SFA in this paper. A brief explanation of this method follows; for a detailed exposition of SFA and DEA, see Coelli et al.[35] and Kumbhakar and Lovell[36].

SFA estimates a producer's (in the following we will refer to farms) TE by measuring the distance between its observed input–output combination and the best-feasible input–output combination that it could obtain if it were located on the so-called '*production frontier*'. This production frontier defines the highest possible amount of output that can be obtained from any given amount of input ('*output oriented approach*'). Alternatively, the frontier can be defined in terms of the lowest amount of input that is required to produce a given amount of output ('*input oriented*'). In agriculture, where many key inputs such as land, machinery and buildings are given, at least in the short run, the output-oriented approach is often favored.

Deviations between individual farms' observed output and the frontier output could occur for two reasons. The first reason is *technical inefficiency*—the fact that some farmers are more successful than others at putting the inputs at their disposal to the best possible use, i.e., at feeding cows to produce milk or at tending crops. The second reason is due to *stochastic effects* such as weather, luck or measurement errors. It is important to allow both components in order to avoid confounding the technical inefficiency estimates with purely random errors. The SFA approach accomodates this requirement by using a composed error structure, where one one-sided error component captures technical inefficiency, and a second error, symmetric error component, captures random deviations from the frontier. This basic SFA model is illustrated in Figure 2.

Due to the presence of stochastic effects, the production frontier is refered to as '*stochastic*', and for Farms A and B with input levels $x_a$ and $x_b$ we see how its observed output deviates from its '*frontier output*' $y_b^*$ due to inefficiency and stochastic effects. Note that with a large positive stochastic effect $v$ the observed output can lie above the frontier. Econometric SFA techniques estimate the location of the stochastic production frontier from data on the input use and output production of a set of farms, and decompose the deviation of each individual farm's actual production from the frontier into its inefficiency and stochastic components. Of course, in reality, farms use many inputs to produce more than one output, but the SFA model depicted in Figure 2[37,38] can be generalized to deal with the multi-input/multi-output case.

Formally, this generalized SFA model that we employ can be written as

$$y_{it} = f(x_{jit};\beta)^* \exp\{w_{it}\} \quad \text{with} \quad w_{it} = v_{it} - u_{it}, \qquad (1)$$

$$y_{it} = f(x_{jit};\beta)^* \exp\{v_{it} - u_{it}\}. \qquad (2)$$

In these equations, $y_{it}$ is the output of farm $i$ in year $t$ measured as its total agricultural revenue, and $x_{jit}$ measures the use of input $j$ by farm $i$ in year $t$. In our application, there are $j = 5$ inputs:

$x_1$: intermediate inputs (such as feed and electricity);

$x_2$: other operating expenses (such as repairs to buildings and machinery, insurance premia and fees for membership in professional organizations);

$x_3$: the depreciation of fixed assets (buildings and machinery);

$x_4$: labor, measured as total labor in agricultural working units per year; and

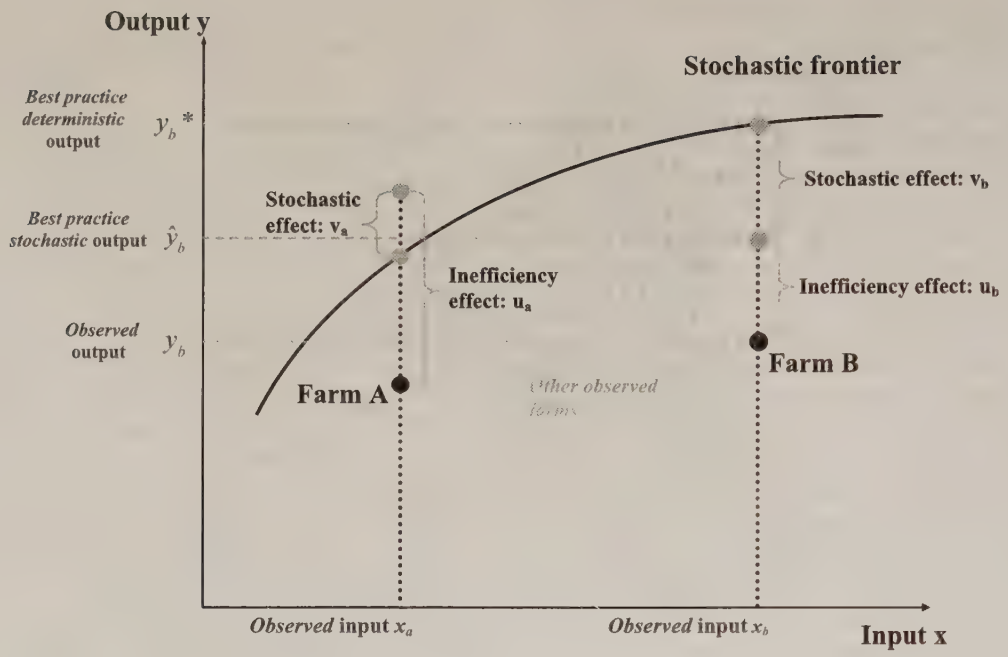


**Figure 2.** The estimation of the stochastic frontier model. Source: Own presentation based on Coelli et al.[35] p. 244.

$x_5$: land, measured as utilized agricultural area in hectares.

β denotes a vector of parameters to be estimated that describe the specific mathematical form of the function $f(\cdot)$ that relates input use $x_{jit}$ to output $y_{it}$. In most applications of the SFA method, including ours, a functional form called the 'translog' is used. The translog function broadly satisfies the requirements of economic-production theory but is at the same time sufficiently flexible to allow the data to 'speak for themselves' and permit subsequent testing of conditions such as decreasing marginal returns to individual inputs and whether or not returns to scale in production are constant.

Finally, the combined error-term $w_{it}$ in Equation 1 is split into the two components introduced above, as depicted in Equation 2. $v_{it}$ captures stochastic effects for farm $i$ in year $t$ and is assumed to be normally distributed. $u_{it}$ measures the inefficiency of farm $i$ in year $t$; $u_{it}$ is greater than or equal to zero because a farm's input–output combination can only lie below the production frontier (in which case $u_{it}$ is positive and is a measure of the farm's inefficiency) or exactly on the frontier (in which case $u_{it}$ equals zero and the farm is perfectly efficient). The non-negative part of a normal distribution $u_{it}$ is frequently used in SFA applications, and we follow this practice. Hence, denoting by μ the mean and by $\sigma_u^2$ the variance of the underlying normal distribution, the inefficiency related errors $u_{it}$ are modeled by truncation from below as $u_{it} \sim N^+(\mu, \sigma_u^2)$. The mean and the variance of the truncated distribution $u_{it}$ itself will differ from μ and $\sigma_u^2$, the mean is

$$E(u) = \mu + \sigma \frac{\varphi(-\mu/\sigma)}{1-\Phi(-\mu/\sigma)}, \tag{3}$$

and the variance is

$$\mathrm{VAR}(u) = \sigma^2 \left(1 - \frac{\varphi(-\mu/\sigma)}{1-\Phi(-\mu/\sigma)} \left(\frac{\varphi(-\mu/\sigma)}{1-\Phi(-\mu/\sigma)}\right) + \mu/\sigma\right). \tag{4}$$

$\mathrm{TE}_{it}$ is defined as the ratio of empirically observed output $\hat{y}_{it}$ to the maximum feasible output $y_{it}^{\max} = f(x_{jit};\beta)\exp(v_{it})$ so that

$$\mathrm{TE}_{it} = \frac{\hat{y}_{it}}{y_{it}^{\max}} = \exp(-u_{it}). \tag{5}$$

Jondrow et al.[39] outline the econometric techniques that are used to decompose the error term $w_{it}$ into its inefficiency and stochastic components $u_{it}$ and $v_{it}$, respectively[39]. Econometric estimation of the model in Equation 1 hinges on the assumption that both these error components are homoscedastic, in other words they have constant variances[36]. However, this may not be the case. In particular, the inefficiency term $u_{it}$ is likely to vary with the size of a farm, since a farm with a large input endowment (and therefore high output capacity) might exhibit substantially more variation, and therefore more scope for inefficiency than a smaller farm[40]. To account for this possibility, we estimate the following modification of the SFA model[10,40].

$$\sigma_{u_{it}} = \exp(x_{jit}\rho_j). \tag{6}$$

In Equation 6, the variance of the inefficiency term is a function of the $j$ inputs $x_{it}$ for farm $i$ in year $t$ introduced above, and $\rho_j$ is a corresponding set of $j$ parameters to be estimated. A positive estimate of $\rho_j$ implies that the corresponding input $x_j$ leads to a larger variance of the inefficiency term. This in turn indicates that the input has a negative impact on TE.

**Table 1.** Description of the variables in the production function.

| Variable description | | Unit | Minimum | Mean | Maximum | Standard deviation |
|---|---|---|---|---|---|---|
| Sum of agricultural revenue | $y$ | € | 586 | 91,752 | 931,220 | 72,162 |
| Intermediate inputs | $x_1$ | € | 915 | 38,845 | 527,230 | 41,135 |
| Other operating expenses | $x_2$ | € | 902 | 21,037 | 187,050 | 15,714 |
| Depreciation | $x_3$ | € | 140 | 22,622 | 234,620 | 16,336 |
| Labor | $x_4$ | AWU/year[1] | 0.13 | 1.81 | 15.16 | 1.09 |
| Land | $x_5$ | hectares | 0.70 | 65.75 | 1,041.80 | 82.96 |

Source: Own calculations.
[1] AWU, standardized agricultural work units.

The influence of additional potential determinants of TE can be estimated in terms of the parameter $\mu$ of the truncated normal distribution[9] as specified in Equation 7:

$$\mu_{it} = z_{kit}\delta_k + e_{it}, \quad (7)$$

where $z_{kit}$ is a setof $k$ determinants of TE for farm $i$ in period $t$, and $\delta_k$ is a corresponding set of $k$ parameters to be estimated. A positive estimate of $\delta_k$ indicates that the corresponding variable $z_k$ increases the distance between a farm's input/output combination and the production frontier, and therefore has a negative impact on TE.

In order to see the regional distribution of efficiency scores, we applied a standard cluster analysis with the 'partitioning around medoids (PAM)' algorithm[41]. The idea is to cluster the farms according to their geographical location information, which are given as 'Universal Transverse Mercator (UTM)' values. We used the $x$ and $y$ values of the UTM value, which are the north–south- and the west–east-coordinates, as input to the algorithm to produce 15 regional cluster.

## Data

We use accounting data for organic pasture farms in Germany from 1994/1995 to 2004/2005. Pasture farms are defined as '*specialized pasture farms*' that derive more than 66% of their agricultural revenue from products based on pasture, or '*mixed farms with a focus on pature farming*' that derive at least 33% of their revenue from such products. While some production of sheep, goats and suckler cows is pasture-based, the majority of the pasture in Germany is used in dairy production, and milk is the most important pasture-based output of the farms in our sample. The data were collected according to the agricultural accounting standard of the German Federal Ministry for Nutrition, Agriculture, and Consumer Protection, and made available by LAND DATA, which is a private firm that offers professional services with respect to bookkeeping and official tax declarations for farms. The dataset, therefore, covers farms that use the services provided by LAND DATA for their accounting, and hence is not the product of a representative sampling effort.

Since organic farming in Germany has been expanding rapidly over the past roughly 15 years, the characteristics of a representative sample have been changing as well. While the regional distribution of the farms in the sample is similar to that reported in official statistics[42], two types of organic pasture farms are under represented. The first group comprises very small farms. Many of these farms have no detailed accounting, and most are part-time and can be considered non-commercial or hobby enterprises. These farms will presumably be of below-average efficiency (for example, because they have no accounting that could be used as a management tool), but their omission from the dataset is not critical because their share of pasture production in Germany is small, and they are not the farms that will shape the future of organic pasture farming in Germany. The other group of farms that is under represented are large former collective farms in eastern Germany that have converted to organic pasture production. Some of these farms are large enough to have their own accounting specialists and, hence, they do not engage the services of LAND DATA. Despite the fact that this type of farm is under represented, we do have a number of east German farms (62) in our sample, and there is no reason to expect any systematic differences between these farms and those that are not in the sample. Overall, we are confident that the sample provides a robust basis for analyzing the determinants of TE in German organic pasture farming, for a detailed discussion of the sample and its representativity, see Lakner[42].

The subset of this data that we analyze consists of an unbalanced panel with 1717 observations from $i = 396$ organic pasture farms over $t = 11$ years. Observations with obvious inconsistencies or missing values were deleted from the sample, and monetary variables were deflated using the official price indices for agricultural products and for agricultural inputs that are provided annually by the German Federal Office for Statistics[22]. All input variables were normalized by dividing by sample means, except for the linear trend that enters in deviations from the sample mean.Table 1 describes the input and output variables in the production function.

We use the following set of $z$ variables as potential determinants of TE in the estimation of Equation 7 (variable names in *italics*, descriptive statistics in Table 2):

**Management capacity and human capital.** Farmers with *No Agricultural Education* from a university, technical college or agricultural school are expected to be less technically efficient. However, earlier studies find that

**Table 2.** Variables used as determinants of TE.

| Variable | Unit | Minimum | Mean | Maximum | Standard deviation |
|---|---|---|---|---|---|
| **Management capacities and human capital** | | | | | |
| In conversion | 0/1 | 0 | 0.1 | 1 | 0.3 |
| No agricultural education of the farmer | 0/1 | 0 | 0.1 | 1 | 0.3 |
| Age of the farmer | Years | 20 | 43.5 | 70 | 8.4 |
| Expenses for legal advice | € | 0 | 58.3 | 6572 | 329.6 |
| **Farm structure and available natural resources** | | | | | |
| Soil quality | EMZ/ha | 25 | 3344.5 | 9877 | 1256.0 |
| Equity share | % | 0 | 77.8 | 100 | 24.0 |
| Sum of land rent | € | 0 | 6428.4 | 111,290 | 9242.1 |
| Volume of milk quota | kg/yr | 0 | 8645.6 | 307,430 | 19,135.0 |
| Intensity of livestock production | VE/ha[1] | 0 | 1.0 | 2.4 | 0.4 |
| Pasture share | % | 0 | 61.2 | 100 | 28.1 |
| **Institutional choice** (legal form, tax options) | | | | | |
| Legal status = *GbR* (civil law association according to the German civil code) | 0/1 | 0 | 0.2 | 1 | 0.4 |
| Part-time farm | 0/1 | 0 | 0.1 | 1 | 0.3 |
| **Market orientation and policy support** | | | | | |
| Volume of subsidies | € | 0 | 12,588 | 163,990 | 10,659 |
| **Regional variables** | | | | | |
| Share of organic farmers | % | 0.3 | 4.2 | 16.4 | 2.8 |
| Eastern Germany[2] | 0/1 | 0 | 0.07 | 1 | 0.3 |
| Northern Germany | 0/1 | 0 | 0.05 | 1 | 0.2 |
| Western Germany | 0/1 | 0 | 0.08 | 1 | 0.3 |
| Share of votes for the Green Party | % | 2.4 | 6.6 | 18.4 | 2.2 |
| Distance to the nearest organic dairy | km | 4.9 | 38.4 | 154.4 | 26.3 |

Source: Own calculations.
[1] VE = 'Vieheinheiten', which are standardized animal units. [2] The reference regional dummy variable is for southern Germany.

education has no impact on the success of organic farms[18,43], and it may be that the education provided in most Germany universities, technical colleges and agricultural schools to date does not address the specific needs of organic farming. A farm manager's *Age* captures the impact of both education and experience. The literature includes studies with positive[44] and negative[10] impacts of age on TE. The impact of *Expenses for Legal Advice* is ambiguous: taking legal advice could lead to more efficient enterprise organization and management, but it might also reflect entanglement in disputes that bind resources and distract from other management tasks. As discussed above, farms in *Conversion* are expected to be less technically efficient.

**Farm structure and resources.** *Soil Quality* (EMZ/ha) is expected to exert a positive impact on TE[18]. Soil quality is measured in '*Ertragsmesszahl*' (*EMZ*) according to the German soil classification system. EMZ/ha ranges from 25 (poorest) to 10,000 (best) and reflects characteristics such as soil texture (percentage of clay, silt and sand) and water retention capacity as well as climatic aspects such as temperature patterns. A high *Share of Pasture Area* in total agricultural area and a high *Intensity of Livestock Production* (measured in animal-units per hectare) are indicators of specialization that both can be expected to lead to higher efficiency. Financial management variables such as *Share of Equity* could affect TE in either direction[13], depending on whether agency theory (monitoring) or credit-evaluation issues (lender aversion to risky credits) dominate. In the European Union, the milk production is limited by a *milk quota* that allows farmers to deliver milk to the dairies and that was originally distributed to the farms in 1984. Since 2004 quota can be traded within different German regions, which makes allocation of quota to growing farms possible, but which is also a cost factor. The *Volume of Milk Quota* owned by the farm might therefore have a positive impact on efficiency, since farms that own quota do not have to rent it. Furthermore, this variable captures the difference between milk producers and other pasture farms in the sample that do not produce milk (suckler cows, sheep and goats). Farmers who rent land are obliged to pay a market price for the use of this land, whereas those who farm their own land can hide a lack of efficiency behind low-imputed rental payments for this land. Hence, we expect that as a farm's *Total Expenditure for Land Rental* increases, so will its TE.

**Institutional choices.** Farms with the legal status of a *Gesellschaft bürgerlichen Rechts* (*GbR* or 'civil law association' according to the German Civil Law section 705) face high set-up costs and internal transaction costs. The *GbR* is frequently observed when two or more farmers establish a partnerhsip. We therefore expect that the establishment of a *GbR* will be associated with strategic

**Table 3.** Coefficient estimates for the production function.

| | Coefficient | *t*-value | | Coefficient | *t*-value |
|---|---|---|---|---|---|
| $\beta_0$ | 0.3250 | 10.10 | $\beta_{14}$ | −0.0402 | −0.93 |
| $\beta_1$ | 0.3327 | 14.90 | $\beta_{15}$ | −0.0086 | −0.30 |
| $\beta_2$ | 0.1777 | 7.18 | $\beta_{1t}$ | 0.0008 | 0.17 |
| $\beta_3$ | 0.0798 | 4.25 | $\beta_{23}$ | 0.0705 | 1.94 |
| $\beta_4$ | 0.1832 | 6.15 | $\beta_{24}$ | −0.0339 | −0.78 |
| $\beta_5$ | 0.1772 | 5.46 | $\beta_{25}$ | 0.1158 | 3.10 |
| $\beta_t$ | 0.0121 | 4.30 | $\beta_{2t}$ | −0.0005 | −0.11 |
| $\beta_{11}$ | 0.2179 | 8.40 | $\beta_{34}$ | −0.1459 | −4.48 |
| $\beta_{22}$ | 0.0004 | 0.01 | $\beta_{35}$ | −0.0149 | −0.40 |
| $\beta_{33}$ | 0.0666 | 1.70 | $\beta_{3t}$ | −0.0014 | −0.34 |
| $\beta_{44}$ | 0.0389 | 0.75 | $\beta_{45}$ | 0.0895 | 1.73 |
| $\beta_{55}$ | −0.0646 | −1.65 | $\beta_{4t}$ | 0.0040 | 0.69 |
| $\beta_{tt}$ | −0.0067 | −4.20 | $\beta_{5t}$ | −0.0028 | −0.50 |
| $\beta_{12}$ | −0.1064 | −3.29 | ln $\sigma_v$ | −2.0170 | −36.90 |
| $\beta_{13}$ | −0.0504 | −1.31 | | | |

Source: Own calculations.

restructuring efforts and, thus, increased TE. *Part-Time Farms* might be less efficient because a part of the family income is derived from other activities and the family therefore relies less on the efficiency of farm operations.

**Market orientation/policy support.** The impact of the *Volume of Subsidies* that a farm receives for organic production is ambiguous. Organic farms receive per hectare premium payments from EU agri-environmental programs (see EU-VO 2078/92 and EU-VO 1257/99). Many publications point to the negative impact that subsidies have on efficiency by weakening managerial efforts[45,46]. However, subsidies can lead to increased investment and thus efficiency, as for example reported in a study on dairy farms in Denmark, Sweden and Finland[47]. Hence, we formulate no expectations for this variable.

**Location.** The regional variables discussed above (*Regional Share of Organic Farmers* and *Local Election Results of the Green Party*) are provided by the German Federal Office for Statistics that provides data at the county level every 2 years[23]. Election results for the Green Party are taken from national elections. The results of state or local elections might better depict local voter preferences, but voter participation is much lower and more variable in these elections. The *Distance to Nearest Organic Dairy* was calculated using ArcView 10. We also include a set of dummy variables to capture regional differences in production systems in Germany. Analysis by the International Farm Comparison Network (IFCN) shows that the large milk producers that emerged from the former collective farms in eastern Germany are very competitive internationally[48]. Hence, we expect the dummy for *eastern Germany* (the omitted reference dummy is for *southern Germany*; dummies for *northern* and *western Germany* are also included) to have a significant positive impact on TE.

We estimated the model with *Oxmetrics 6.0* and the program package *sfamb for ox*[49].

## Results and Discussion

The estimation results for the *production function model* (Eqn 2) are presented in Table 3. Most of the estimated coefficients are significantly different from zero at the 5% level. Because all variables have been normalized by their respective sample mean prior to taking logarithms, the first-order estimates $\beta_j$ can be interpreted as partial production elasticities at the sample mean that shows how much the output would increase in percentage terms if the use of the respective input was increased by 1%. Of the five inputs, intermediate inputs have the largest partial production elasticitiy ($\beta_1 = 0.33$). The estimated elasticity of labor ($\beta_4 = 0.18$) is larger than that found for conventional dairy farms[10]. This is plausible since the labor share is higher on organic than on conventional farms. The other inputs play less important roles. The estimated annual rate of technological progress (indicated by $\beta_t$) is 1.2%.

The estimated ratio of the inefficiency variance to the total error variance, $\gamma = \sigma_u^2/\sigma_u^2+\sigma_v^2$, is 0.95. Hence, most of the variation in the composite error term is due to the inefficiency component. The average TE score is 0.74, with a standard deviation of 0.21; this indicates that the average farm in the sample is producing 74% of the maximum feasible output for its level of input use.

Table 4 presents the results of selected *specification tests* for the production function model with technical effects.

$H^1$ tests the hypothesis that every farm in the sample is fully efficient and that the introduction of an 'inefficiency term' is therefore not justified. This hypothesis is rejected, which indicates that inefficiency effects play a significant role. The hypothesis of linear homogeneity ($H^2$) is rejected, which means that the assumption of globally constant returns to scale is not supported by the data. The mean scale elasticity is 0.93 and we find decreasing returns to scale on about 79.1% of the farms, meaning that if the use of all inputs was increased by some proportion $X$ on each of these farms, their output would increase by less than proportion $X$. This result contrasts with results for dairy farms in the USA, which show increasing returns to scale for both organic and conventional farms[14]. The rejection of $H^3$ indicates that the $z$ variables in Equation 7 have a significant joint impact on technical efficienty, and the rejection of $H^4$ indicates that the subset of $z$ variables that capture location effects also has a significant joint impact on TE. Other specification tests (available from the authors) favor the translog functional form of the frontier production function over the simpler Cobb–Douglas form. Overall, the results in Table 4 support the specification of the SFA model proposed above, including variables that capture location effects as determinants of TE.

The results from the 'heteroscedasticity model' (the parameter estimates for the $x$-variables presented in Eqn 6) show that efficiency is significantly affected by different factor endowments on the farms. For example, farms with higher levels of intermediate inputs, labor and land use

**Table 4.** Results of tests for model quality.

| Test | Null hypothesis | Test value | Critical value[1] | Result |
|---|---|---|---|---|
| $H^1$: no inefficiency | $\gamma = 0;\ \rho = 0;\ \delta = 0$ | 1,312.70 | 30.81 | Reject |
| $H^2$: linear homogeneity | $\Sigma\beta_j = 1;\ \Sigma\Sigma\ \beta_{jk} = 0$ | 33.06 | 12.59 | Reject |
| $H^3$: no technical effects | $\delta_0 = \delta_1 = \ldots = \delta_{19} = 0$ | 560.29 | 31.41 | Reject |
| $H^4$: no location effects | $\delta_{14} = \delta_{15} = \ldots = \delta_{19} = 0$ | 53.13 | 12.59 | Reject |

Source: Own calculations.
[1] Critical value for $H^1$ from Kodde and Palm[50]; the test statistics for $H^2$ through $H^4$ follow $\chi^2$ distributions. The critical values are significant at the 5% level of error probability; all test values are significant at the 1% level.

**Table 5.** Estimated coefficients for the technical effects model.

| Variable | Parameter | Coefficient | *t*-value | Marginal effect |
|---|---|---|---|---|
| Constant | $\delta_0$ | −0.073 | −0.35 | . |
| In conversion | $\delta_1$ | 0.133*** | 2.58 | −0.004 |
| No agricultural eduction of the farmer | $\delta_2$ | −0.069 | −0.72 | 0.002 |
| Age of the farmer | $\delta_3$ | 0.0004 | 0.25 | −0.000 |
| Expenses for legal advice | $\delta_4$ | 0.008* | 1.82 | −0.0002 |
| Soil quality | $\delta_5$ | −0.100** | −2.52 | 0.003 |
| Equity share | $\delta_6$ | −0.001 | −0.18 | 0.000 |
| Sum of land rent | $\delta_7$ | −0.031*** | −3.87 | 0.001 |
| Volume of milk quota | $\delta_8$ | −0.012*** | −2.87 | 0.0004 |
| Intensity of livestock production | $\delta_9$ | −0.441*** | −4.75 | 0.014 |
| Pasture share | $\delta_{10}$ | −0.068** | −2.10 | 0.002 |
| Legal status = *GbR* | $\delta_{11}$ | −0.171** | −2.04 | 0.003 |
| Part-time farm | $\delta_{12}$ | −0.085 | −1.26 | 0.002 |
| Volume of subsidies | $\delta_{13}$ | 0.015** | 2.32 | −0.001 |
| Regional variables | | | | |
| Regional share organic farmers | $\delta_{14}$ | −0.130** | −1.98 | 0.004 |
| Eastern Germany | $\delta_{15}$ | 0.221** | 2.27 | −0.004 |
| Northern Germany | $\delta_{16}$ | −0.363*** | −2.73 | 0.006 |
| Western Germany | $\delta_{17}$ | −0.346*** | −3.27 | 0.006 |
| Share of votes for the Green Party | $\delta_{18}$ | −0.099* | −1.76 | 0.003 |
| Distance to the nearest organic dairy | $\delta_{19}$ | 0.108** | 2.52 | −0.003 |

Source: Own calculations.
*Estimated results with 10% error-probability, **estimated results with 5% error probability, ***estimated results with 1% error probability.

tend to have a smaller $\sigma_u$, in other words to be less inefficient.

Table 5 presents the parameter estimates and marginal effects for the *determinants of TE* (the $z$ variables in Eqn 7). Marginal effects measure the change in TE in percentage points due to a small change in the corresponding determinant[51]. These are estimated for the continuous variables by numerical differentiation at the sample mean, and for dummy variables, they measure the impact on TE of a unit change (from zero to one) in the corresponding variable.

Among the management variables, we find that farms in *Conversion* to organic production are less technically efficient than converted organic farms, a result that we discuss in more detail below. Farmers with *No Agricultural Education* are neither more nor less technically efficient than others, and *Age* also has no significant influence on efficiency. The first of these results confirms the results from other studies in organic farming[14,18,43]. It suggests that agricultural education in German universities, technical colleges and agricultural schools has, at least to date, not imparted knowledge and skills that contribute to technically efficient organic farming. *Expenses for Legal Advice* do not have a significant impact on TE.

As a farm's *Soil Quality*, *Pasture Share* and *Volume of Milk Quota* increase, so does its TE. A farm's *Equity Share* has no significant impact on its TE. As expected, the *Total Expenditure for Land Rent* has a positive impact on TE, and this impact is highly significant. Mayen et al.[14] argue that renters of land might have lower incentives to use rented land efficiently. This might be true in the USA and in general for markets where rental prices of land are low. But as competition on the landmarket and rental prices for land increase (as in west Germany), the incentives to use land efficiently should increase as well, resulting in higher-efficiency scores (as estimated here).

The *Intensity of Livestock Production* also has a strong influence on TE, with a 1% increase in intensity boosting efficiency by nearly 1.4%. As expected, farms that have the

**Figure 3.** Average TE in 15 clusters of organic pasture farms in Germany. Source: Own presentation.

status of a *GbR* perform better than those that have adopted other legal forms. *Part-Time Farms* are more efficient, but this effect is not statistically significant. Farms that receive larger *Volumes of Subsidies* have lower-efficiency scores, although the impact is small. This may be evidence that subsidies for organic farms, which largely take the form of payments for agri-environmental services, lead farms to pursue other activities that detract from the core pursuit of efficient pasture-based production.

Most of the coefficients for variables that capture *location effects* are significant at the 5% level. There are distinct regional differences in TE. Compared with the reference region (southern Germany), the pasture farms in western and northern Germany are more efficient, other things being equal, while the pasture farms in eastern Germany are less efficient. This contradicts the findings of other studies[48], which find that organic milk producers are more competitive in eastern Germany than in other regions. However, the magnitude of the differences estimated here is small; farms in eastern Germany are on average 0.4% less efficient than those in southern Germany, and 1.0% less efficient than those in northern and western Germany. Furthermore, TE is only one component of competitiveness[52]; it could be that organic pasture farms

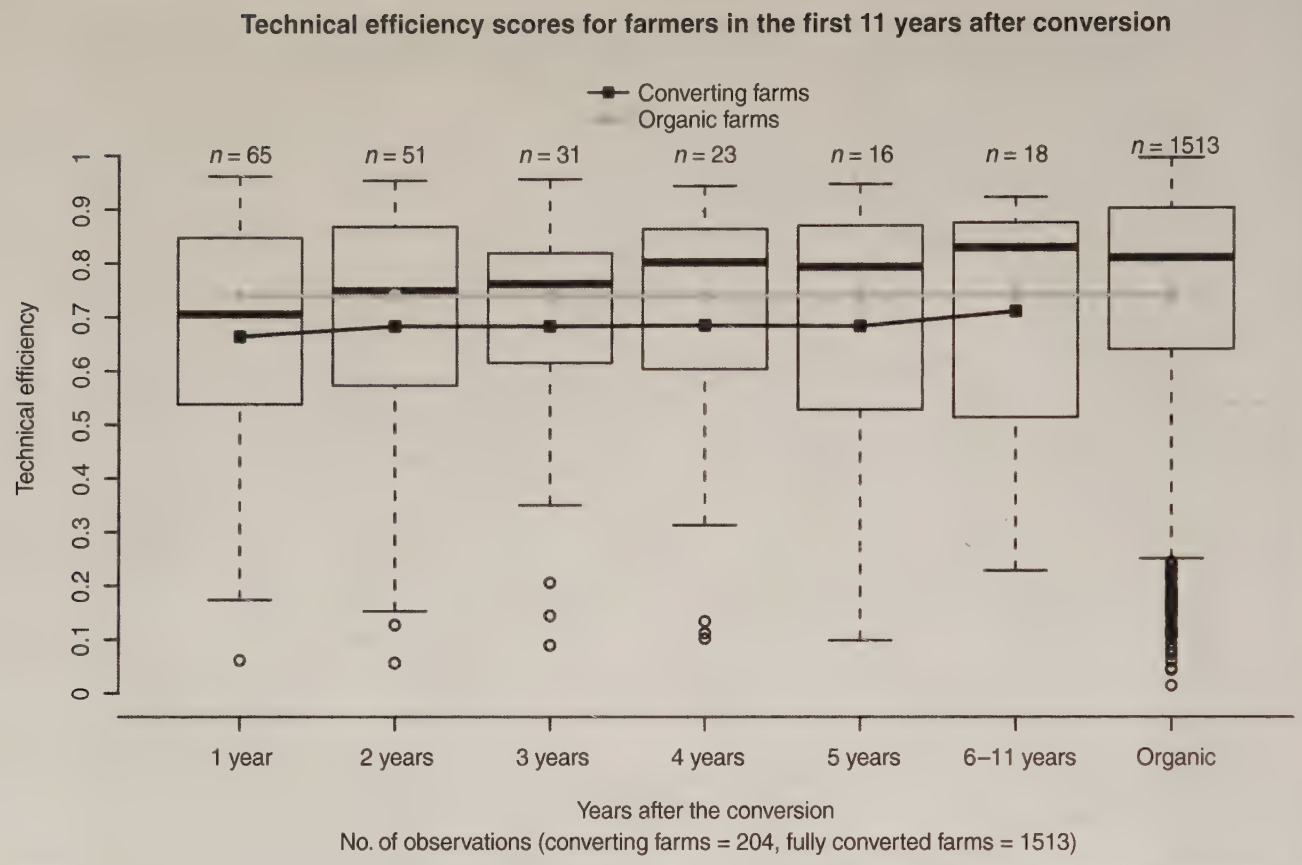


**Figure 4.** TE scores for farmers in the first 11 years after conversion. Source: Own calculations. Note that in the figure the efficiency of the farms 'in conversion' ($n = 204$, *in black*) is grouped according to the year after their conversion (i.e., 1 year, 2 years, . . .), whereas the performance of fully converted organic farms ($n = 1513$, *in grey*) is grouped altogether, denoted as 'organic'. The boxplots 1–6 are based on the data of the farms in the 11 years after the conversion (i.e., $n = 204$), boxplot 7 depicts the distribution of the fully converted organic farms (i.e., $n = 1513$).

in eastern Germany are able to compensate for their slightly lower TE by being larger and thus being able to negotiate better prices for their input purchases and output sales.

The significant negative coefficient on the variable *Regional Share of Organic Farmers* is evidence in favor of localization economies in organic pasture farming. The fact that TE decreases with increasing *Distance to the Nearest Organic Dairy* and increases with the *Share of Votes for the Green Party* suggests that urbanization economies also play a role. All other things being equal, a 1 km reduction in the distance to the nearest organic dairy is accompanied by a 0.3% increase in TE, while a 1% increase in the local Green election result increases TE by 0.3%.

Regional cluster (modeled as described in subsection 'Data') reveals that TE varies substantially across regions. Figure 3 shows 15 regional clusters with their mean TE score.

Especially in southern Germany we can find very efficient clusters ('Allgäu' with 0.8, Bodensee with 0.84 and the region of Munich with 0.82 average TE), whereas the clusters in east Germany show a relatively low performance (Saxony with 0.54 and Brandenburg with 0.33 average TE). Interestingly, especially the southern region with a very high share of organic farms in the south perform very well, whereas in the north-eastern regions, where we can also find high shares of organic farms (see Fig. 1), the efficiency performance is low.

Returning to the impact of *conversion* from conventional to organic farming, Figure 4 illustrates the development of TE over the course of conversion to organic farming. Figure 4 shows that converting farms have lower levels of average TE than converted farms, but that this gap closes over time. These results are as expected and coincide with the findings of Lohr and Park[19], who find lower TE scores in converting farms with less than 5 years of experience. Note that lower efficiency during conversion might be partly price-driven because the output of farms in conversion does not qualify as 'fully' organic, i.e., it has to be labeled differently and generally sells for lower prices. In a sample of organic farmers in Austria that had converted back to conventional production after 5 years, 58% reported converting back because they had only received conventional prices for their organic output[53]. Since we measure output in monetary terms in our application, lower prices could be showing up as reduced efficiency. However, closer examination of the data reveals no significant price differences between the converting and converted farms

in our sample, which suggests that the lower TE of converting farms is not price driven.

The boxplots 1–6 in Figure 4 provide information on the distribution of efficiency scores for farms in the first 11 years after the conversion ($n = 204$), boxplot 7 shows the respective distribution for organic farms that are already converted ($n = 1513$). They show that in most years the distribution of TE is wider in the group of the converting farms than in the reference group of converted organic farms. They show that although converting farms are less technically efficient on average, the range of efficiencies is large and highly skewed toward lower-efficiency scores. The same is true of the efficiency distributions for converted farms.

## Conclusions

We studied the TE of organic pasture farms in Germany using an unbalanced panel with 1717 observation from 396 farms over 11 years. TE is estimated using a stochastic translog production frontier that allows for the simultaneous estimation of the impact of specific variables on technical inefficiency. Along with the factors such as a farmer's age and education that are commonly used to explain TE in agriculture, we explicitly include variables that capture localization and urbanization effects.

The empirical results confirm that localization and urbanization effects do influence the TE of organic pasture farms in Germany. Specifically, TE increases with the regional share of organic farmers, with decreasing distance to the nearest organic milk processing plant, and with the regional share of votes for the 'Green Party' in federal elections. This suggests that future studies of TE in organic agriculture and also agriculture in general should take localization and urbanization effects into account or risk mis-specification. Our results also confirm that the conversion status of an organic farm can have a significant impact on its efficiency. As long as organic farming expands, samples of organic farms will include farms at different stages of the conversion process, and the results here suggest that empirical analysis should explicitly account for this.

The results of our analysis also have implications for policy. We find that organic farmers who have received no specific agricultural education are not less efficient than other organic farmers. While our binary measure of specific agricultural education is admittedly simple, this result matches the results of other studies and supports the supposition of many proponents of organic farming in Germany that agricultural education to date in Germany (which is almost exclusively publicly funded) has not been tailored to the needs of organic farming.

We also find that TE falls significantly with the volume of subsidies received, which was not expected *ex ante*. This might be evidence that the agri-environmental activites for which organic pasture farmers receive special subsidies are detracting from the efficient management of their core production activities. Of course, if farmers are volunteering to participate in these schemes, then presumably the subsidies that they are receiving in return must compensate for any resulting loss in efficiency. Nevertheless, if this explanation holds, German policy-makers might want to reconsider the types of subsidies that they have been providing to organic farming and design schemes that increase rather than decrease efficiency. However, this result might also reflect that the less efficient farmers are more likely to opt for participation in subsidy programs. Note that some of these programs require delivery of concrete environmental benefits, which from a broader perspective might compensate for lower TE. This links to the topic of 'environmental efficiency'[54], where more research seems to be necessary.

Finally, we find that farms that are converting to organic production are less efficient on average than converted farms, although the difference in efficiency between these groups of farms falls as conversion progresses. This finding could be used to justify temporary and phased support for converting farms[55]. In most German federal states, the special premia for farms converting to organic production were abolished in 2006/2007. However, our results also show that the distribution of TE among converting organic pasture farms is wide and skewed. Therefore, it would be difficult to determine an appropriate level of support and in particular avoid overcompensation of converting farms that are relatively efficient. How to dose and target support and what instruments should be used to provide it (e.g., per hectare payments, price supports and extension) are topics for further research.

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# Ecological impact of wheat and spelt production under industrial and alternative farming systems

Martina Bavec[1], Michael Narodoslawsky[2], Franc Bavec[1], and Matjaž Turinek[1,*]

[1]Institute for Organic Farming, Faculty of Agriculture and Life Sciences, University of Maribor, Pivola 10, SI-2311 Hoče, Slovenia.
[2]Institute for Resource Efficient and Sustainable Systems, Graz University of Technology, Inffeldgasse 21 B, A-8010 Graz, Austria.
*Corresponding author: matjaz.turinek@uni-mb.si

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## Abstract

The Industrial Revolution and intensification of agriculture have, in some cases, led to economic activities that profoundly influenced the ecosystem to the point where environmental stability and geographic political security are jeopardized. The uncertainty about oil reserves, rising energy prices and the threat of harmful climate change effects has intensified the search for alternative farming systems that reduce negative environmental impact. This study reports the ecological impact of conventional (CON), integrated (INT), organic (ORG) and biodynamic (BD) farming systems calculated from data collected in a field trial at Maribor, Slovenia, and interpreted using the SPIonExcel tool. This tool is a member of the ecological footprint family and describes the area necessary to embed a human activity sustainably into the ecosphere. Three-year results show a markedly reduced ecological footprint of the ORG and BD systems in production of wheat (*Triticum aestivum* L. 'Antonius') and spelt (*Triticum spelta* L. 'Ebners rotkorn'), mainly due to the absence of external production factors. When yields were also considered, the ORG and BD systems again had a reduced overall footprint per product unit and increased ecological efficiency of production. Thus, ORG and BD farming systems present viable alternatives for reducing the impact of agriculture on environmental degradation and climate change. Nevertheless, room for improvement exists in the area of machinery use in all systems studied and yield improvement in the ORG farming system.

**Key words:** production systems, organic farming, biodynamic farming, ecological footprint, SPI, ecological efficiency of production

## Introduction

The Industrial Revolution and intensification of agriculture have in some cases led, for the first time since the emergence of permanent settlements and agriculture more than 12,000 years ago, to economic activities that profoundly influence the ecosystem to the point where environmental stability and geographic political security are jeopardized[1,2]. Thus, the World Commission on Environment and Development (the Brundtlandt Commission) coined the definition of sustainable development in 1987—it is development that satisfies the needs of current generations without compromising the needs of future generations[3]. In recent years, numerous tools and methods have emerged that are supposed to determine sustainable development on the level of single enterprises[4] as well as on a higher, societal level[5,6]. One of these tools is the environmental or ecological footprint[1]. It aims at estimating the biologically productive area needed to produce materials and energy used by the population of a certain region (city, state and world). The calculated area is compared to the area available to a certain population or individual, called the biocapacity, which presents the productive land and/or water of a region. In cases where the ecological footprint is greater than the biocapacity, human consumption exceeds the natural carrying capacity[7]. Data for the ecological footprint are usually based on statistical data; in the case of agriculture, yearly statistics of individual countries from the Food and Agriculture Organization (FAO) of the United Nations are used. The drawback of such data lies in their inherent inaccuracy, making the footprint less useful for evaluating smaller units, e.g., single farms.

Other tools based on actual and/or real data are more appropriate to evaluate individual production processes.

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**Table 1.** Crop rotation designs for years 2006–2010.

| | Crop rotation design 1 (split 1) | | | Crop rotation design 2 (split 2) | | |
|---|---|---|---|---|---|---|
| **Year** | **Plot 1** | **Plot 2** | **Plot 3** | **Plot 1** | **Plot 2** | **Plot 3** |
| 2006 | Clover–grass mixture | | | Clover–grass mixture | | |
| 2007 | Clover–grass mixture | | | Clover–grass mixture | | |
| 2008 | Wheat | Cabbage | Oil pumpkins | Spelt | Red beet | False flax |
| 2009 | Cabbage | Oil pumpkins | Wheat | Red beet | False flax | Spelt |
| 2010 | Oil pumpkins | Wheat | Cabbage | False flax | Spelt | Red beet |

A framework for applying such evaluation methods is life cycle assessment (LCA), which considers the environmental burden caused by a product, a production process or any activity to provide services[8]. It takes into account the technological processes of all activities along the life cycle, from the provision of basic materials to transportation into and from the production unit to the production process itself and finally the use phase of any product and its safe disposal. It is based on an eco-inventory identifying all material and energy flows exchanged with the environment along the whole life cycle. These flows are then evaluated with an appropriate ecological evaluation method. The result can be interpreted on a per-unit-of-product basis (kg) or equivalent area (ha), where areas used outside of the production unit are included[9]. One drawback of this approach is the limited comparability of the results as they critically depend on the scope of the LCA, which may differ from study to study, even for the same products or services.

Research in the area of the ecological footprint or LCA in agriculture is still developing. In this paper, we will apply ecological evaluation using the LCA framework and compare the production of field crops in different production systems by the Sustainable Process Index® (SPI)[10–13]. This evaluation method has been customized for agriculture, e.g., by introducing an algorithm to account for the impact of seed production. We used experimental data from a systems comparison field trial over 3 years; therefore, results reflect conditions in real-life situations and farming systems. The main question we posed was: how sustainable are the production systems most commonly used today (exemplified by Slovenian wheat and spelt production) and how can they be improved to increase sustainable food production for future generations?

## Materials and Methods

### *Long-term field trial*

The experimental site is located at the University Agricultural Centre of the University of Maribor in Pivola near Hoče (46°28′N, 15°38′E, 282 m a.s.l). The annual mean air temperature of the area is 10.7°C; where the mean monthly minimum is in January at 0.4°C and the average monthly maximum is in July at 20.8°C. Average annual rainfall in the area is around 1000 mm. Sixty 7 m × 10 m experimental field plots were established on a dystric cambisol (deep) [average pH value 5.5 (0.1 M KCl solution), soil soluble P at 0.278 g $kg^{-1}$ and soil soluble K at 0.255 g $kg^{-1}$ in ploughing soil layer], and are maintained within two different five-course crop rotation designs (Table 1). In one rotation there are typical crops for this region [2 years of red clover–grass mixture, winter wheat (*Triticum aestivum* L.), white cabbage (*Brassica oleracea* L. var. *capitata* L. f. *alba*), oil pumpkins (*Cucurbita pepo* var. *styriaca* Greb.]; in the other one there is an alternative crop rotation [2 years of red clover–grass mixture, spelt (*Triticum spelta* L.), red beet (*Beta vulgaris* L.), false flax (*Camelina sativa* L.)]. Since we wanted to compare the same crops in each year of the trial, this resulted in three different combinations of each crop rotation (Table 1). Two years prior to the beginning of the trial a red clover–grass mixture was grown on site and the whole experimental plot was managed according to organic farming standards for 6 years before the trial started in 2007. Four production systems+control plots were arranged in a randomized complete block split-plot design with four replicates, where there were three main plots in each farming system and replicate, which were then split into the two different crop rotations. The farming systems differed mostly in plant protection and fertilization strategies and are defined by the valid legislation and standards (Table 2)—conventional (CON)[14], integrated (INT)[14,15], organic (ORG)[14,16,17], biodynamic (BD)[14,16–18] farming system and control[14] plots, where no fertilization/plant protection was used. Basic soil cultivation, sowing and harvesting dates and methods were identical among experimental plots and were performed on the same dates and in the same manner to adjacent fields (Table 2). Also, the same varieties were used in all farming systems under study (wheat 'Antonius' and spelt 'Ebners Rotkorn'), of conventional origin for CON and INT systems and of organic origin for ORG, BD and control systems.

### *SPIonExcel tool*

The SPI, developed by Krotscheck and Narodoslawsky[11], is based on the assumption that a sustainable economy builds only on solar radiation as natural income. Most natural processes are driven by this income and the Earth's surface acts as the key resource for the conversion of solar radiation into products and services. Global surface area is a limited

**Table 2.** Farming systems under investigation in the field trial and differences between them.

| Production system | Weed management | Pest management | Manure application |
|---|---|---|---|
| CON farming according to the Slovene agriculture act and GAP | Preventive use of herbicides according to GAP, harrowing when needed | Preventive use of pesticides according to GAP | NPK and N mineral fertilizers used according to GAP and nutrient removal estimates |
| INT farming according to Slovene standards for Integrated farming | Use of herbicides according to the rules of INT management (harrowing at least once, no preventive use of herbicides) | Curative use of pesticides according to the rules of INT management (i.e., list of allowed products renewed and published every year by the Ministry of Agriculture) | NPK and N mineral fertilizers used based on soil analysis and nutrient removal estimates |
| ORG farming according to the EC regulation on Organic Farming | Harrowing 1–2 times/season, cover crops after cereals | Crop rotation, harrowing and cover crops | 1.4 LU of rotted cattle manure/ha |
| BD farming according to Demeter International production standards and EC regulation on Organic Farming | Harrowing 1–2 times/season, cover crops after cereals | Crop rotation, harrowing, cover crops, use of BD preparations | 1.4 LU of composted cattle manure/ha with added BD compost preparations |
| Control plots | Harrowing 1–2 times/season | None | None |

BD, biodynamic; EC, European Council; GAP, good agricultural practice; INT, integrated farming; LU, livestock units.

resource in a sustainable economy, and anthropogenic as well as natural processes compete for this resource. Therefore, area needed to embed a certain process sustainably into the ecosphere is a convenient measure for ecological sustainability; the more area a process needs to fulfill a service, the more it 'costs' from an ecological sustainability point of view.

Human activities exert impacts on the environment in different ways. On the one hand they need natural resources (e.g., to provide energy, material means of production like fertilizers, etc.), manpower and area for installations. On the other hand they produce emissions and waste besides the intended goods. Therefore, the SPI includes all these different aspects of ecological pressure on the environment and translates them into surface area required by the process. The conversion of mass and energy flows into area is based on two general 'sustainability principles'[13]:

- Principle 1. Anthropogenic mass flows must not alter global material cycles; as in most global cycles (such as the carbon cycle) the flow to long-term storage compartments is the rate-defining step of these dynamic global systems; flows induced by human activities must be scaled against these flows to long-term stores.
- Principle 2. Anthropogenic mass flows must not alter the quality of local environmental compartments; here the SPI method defines maximum allowable flows to the environment based on the natural (existing) qualities of the compartments and their replenishment rate per unit of area.

Further details of this method would be out of scope for this paper; the basic algorithm used in this work is given below and the method is described in detail elsewhere[13,19].

The software SPIonExcel was developed to bring this methodology into an easily applicable form. It is available on the Internet (http://spionexcel.tugraz.at/) and calculates the ecological footprint of a process, product or service given an eco-inventory summarizing the flows to and from the environment over the life cycle in question.

For this paper, the SPIonExcel tool is modified to increase its applicability for agricultural systems, employing slightly different calculation methods compared to the original method (in particular by taking seed production into account) and using a detailed inventory and database for different production systems.

The modified SPIonExcel tool calculates a total ecological footprint ($A_{tot}$) that is the area necessary to embed the whole life cycle generating a product (e.g., wheat) into the ecosphere. $A_{tot}$ is calculated from 'partial footprints' using the following equation:

$$A_{tot} = A_l + A_{fp} + A_m + A_s \ (m^2), \quad (1)$$

where $A_l$ stands for the footprint of direct land use, $A_{fp}$ for the footprint fertilizer and pesticide, $A_m$ for the footprint derived for machinery use and $A_s$ for the footprint of seed use. Partial footprints were calculated directly from the experimental field trial data, except for the footprints of seed use, which were determined by using Equation 2 from the intermediate footprint (up to seed) of a production system:

$$A_s = \frac{A_l + A_{fp} + A_m}{Y_a} \times S_a \ (m^2), \quad (2)$$

where $Y_a$ stands for quantity (in this case the yield of grain) of a crop produced in 1 year and $S_a$ for the quantity of seed used for crop establishment in a year.

**Table 3.** Sample technological chart for wheat (*T. aestivum* L. 'Antonius') production in the year 2009 with all inputs and machinery use noted.

| Measures | | Production system | | | | | |
|---|---|---|---|---|---|---|---|
| | | Conventional | Integrated | Organic | Biodynamic | Control | |
| Ploughing | 100 HP | 2 | 2 | 2 | 2 | 2 | (h, ***) |
| Seeding | 75 HP | 0.75 | 0.75 | 0.75 | 0.75 | 0.75 | (h, **) |
| Fertilization | 75 HP | 2 | 1.5 | 4 | 4 | / | (h, **) |
| | NPK (7 : 20 : 30) | 400 | 150 | / | / | / | (kg ha$^{-1}$) |
| | N-fertilizer (CAN) | 666 | 406 | / | / | / | (kg ha$^{-1}$) |
| | Potassium salt | / | 100 | / | / | / | (kg ha$^{-1}$) |
| | Stable manure/compost | / | / | 21,450 | 18,000 | / | (kg ha$^{-1}$) |
| | BD preparations | / | / | / | BD 502–507 each 8–10 g | / | |
| Spraying | 50 HP | 1.5 | 1 | / | 2 | / | (h, *) |
| | Herbicide | | / | / | / | / | kg |
| | Boom efekt (glyphosate) | 5 | / | / | / | / | (l ha$^{-1}$) |
| | Stomp (pendimethalin) + Axial (pinoxaden) | 5.2 | 5.2 | / | / | / | (l ha$^{-1}$) |
| | Fungicide | / | / | / | / | / | |
| | Amistar extra (cyproconazole + azoxystrobin) | 1 | 1 | / | / | / | (l ha$^{-1}$) |
| | Insecticide | / | / | / | / | / | |
| | Fastac (alpha-cypermethrin) | 0.12 | 0.12 | / | / | / | (l ha$^{-1}$) |
| | BD preparations | / | / | / | 200 g BD 500 + 15 g BD 501 | / | |
| Harrowing | 75 HP | / | 1 | 1 | 1 | 1 | (h, *) |
| Harvest | 245 HP | 0.7 | 0.7 | 0.7 | 0.7 | 0.7 | (h, **) |
| Yield | | 5800 | 4920 | 2453 | 3560 | 2678 | (kg ha$^{-1}$) |
| Seed used | | 200 | 200 | 200 | 200 | 200 | (kg ha$^{-1}$) |

Intensity of machinery use: *, light; **, normal; ***, heavy; HP, horsepower; BD, biodynamic. A detailed description of BD preparations can be found in Turinek et al.[34].

From the attained total ecological footprint, an additional overall footprint per unit was calculated, namely:

$$a_{tot} = A_{tot}/Y_a \ (m^2\,kg^{-1}), \tag{3}$$

where $a_{tot}$ gives an appraisal of the 'cost' in terms of ecological sustainability of a given product or service by indicating how much surface area is needed to produce one unit of a product, in our case wheat or spelt grain.

The area derived from the above calculation can be related to the surface area that is statistically available to a person in a country, region or area ($a_{inh}$), which can be obtained from statistical data. This relation then represents the fraction of the 'sustainable ecological budget' for a person consuming the product in question provided by a particular production system. This value is called the SPI[13]:

$$SPI = \frac{a_{tot}}{a_{inh}} \times 1000. \tag{4}$$

As the number would be too small if given on a per-kg basis, it was multiplied by 1000 to give it on a per-ton basis and to better visualize differences between production systems.

The efficiency of a production system in providing a good or service is, however, better expressed through the ecological efficiency of production (EEP) calculated in Equation 5. It provides us with the information on how much of a good or service can be produced on 1 ha of surface area in 1 year with the process or system under study, embedding the provision of this good or service totally and sustainably in the ecosphere:

$$EEP = \frac{Y_a}{A_{tot}} \times 10{,}000 \ (kg\,ha^{-1}). \tag{5}$$

## Data used

All work done on the trial in the years 2008, 2009 and 2010 was carefully monitored and recorded. Data collected from the field trial were transformed into tasks done in a system in 1 year and the time needed for those tasks (e.g., ploughing, seeding, harrowing, spraying, etc.). An example is given for wheat production in the year 2009 (Table 3). Because of the nature of the trial, in which not all operations could be done by machine (e.g., spraying and fertilization), real-life operational times were taken from the University Agricultural Centre Farm, where the experiment took place. The footprint was determined for 1 ha of area.

**Table 4.** Yields of wheat and spelt depending on production system and year.

| | Wheat[1] | | Spelt[1] | |
|---|---|---|---|---|
| **Factor** | **Yield (kg ha$^{-1}$)** | **Relative yield[2] (%)** | **Yield (kg ha$^{-1}$)** | **Relative yield[2] (%)** |
| **Production system (PS)** | | | | |
| Control | 2467 ± 207c | 77 | 1807 ± 91b | 83 |
| CON | 4263 ± 469a | 133 | 2260 ± 141ab | 104 |
| INT | 3683 ± 451ab | 115 | 2369 ± 247a | 109 |
| ORG | 2450 ± 263c | 76 | 2039 ± 125ab | 93 |
| BD | 3136 ± 305bc | 99 | 2440 ± 180a | 112 |
| **Year (yr)** | | | | |
| 2008 | 2530 ± 134b | 79 | 1851 ± 57b | 85 |
| 2009 | 3882 ± 431a | 121 | 2550 ± 168a | 117 |
| 2010 | 3186 ± 212ab | 100 | 2149 ± 108b | 98 |
| Average | 3200 | 100 | 2183 | 100 |
| **ANOVA** | | | | |
| PS | *** | | * | |
| Y | *** | | *** | |
| PS × Y | n.s. | | n.s. | |

[1] Wheat yield is given for hulled grain, but spelt yield includes hulls.
[2] Average value of each factor = 100%.
Mean values ± SEs are presented. Different letters indicate statistically significant differences at 95% probability (Duncan test). Levels of significance: n.s., non significant ($P>0.05$); *$P \leq 0.05$; ***$P \leq 0.001$.

### *Statistical analysis*

Data for the yield, $a_{tot}$, SPI and EEP were analyzed by multifactor ANOVA with production system and year as factors using Statgraphics Centurion (Version XV, StatPoint Technologies, Inc., Warrenton, VA) and were followed by least squares means comparisons after Duncan[20]. Values given within the paper are means ± standard error (SE).

## Results and Discussion

### *Yields*

Yields of wheat and spelt varied among production systems and years (Table 4), with no significant interaction between factors. Highest yields of wheat were attained in the CON production system (4263 kg ha$^{-1}$), the lowest in the control and ORG production systems (2467 and 2450 kg ha$^{-1}$, respectively). CON wheat yields were similar to recent average Slovenian wheat yields[21], but lower than those reported by other EU countries. BD and INT systems performed near the average of all farming systems; ORG and control wheat yields tended to be lower (Table 4).

We see a more uniform picture with spelt yields, where differences among production systems are not as accentuated as in the case of wheat. Possibly this is due to the lower breeding modifications of spelt as compared to wheat and the somewhat unresponsive reaction to additional nitrogen fertilizer applications.

For both wheat and spelt, the influence of the production year on yields is significant, where lowest yields were attained in the year 2008 due to significant rain events and thus a delayed harvest (August 7) in that year. Yields in 2009 were above average; 2010, however, gave average yields of both grain crops.

### *Ecological footprint*

The relatively large area appropriated by the CON and INT systems was mostly attributed to mineral fertilizer and pesticide use, while the smaller area appropriated by the ORG and BD systems was mostly due to machinery use (Table 5). For every hectare of CON wheat and spelt production, an additional 52–100 ha of surface area is impacted. The INT system did not perform any better, although it is publicized and advertised as nature-friendlier (compared to the CON system) and as a sustainable agricultural system[15]. Control plots appropriated the least area, which was still seven to eight times greater than the surface area used to plant the crops. In this sense there is great need for improvement in the current agricultural practice and the way we understand, till and work the soil. Furthermore, more efficient machinery and machinery use are a must in order to minimize the impact of agricultural production on the environment.

### *Overall footprint of a product, SPI and EEP*

Results for the $a_{tot}$, SPI and EEP give an even more insightful picture, as yields are taken into the equation (Table 6). For all three parameters, production systems had a significant influence on the attained results for both wheat and spelt, where control, ORG and BD systems outperformed the CON and INT systems. Production year also significantly influenced the $a_{tot}$ and SPI for wheat production. Moreover, the interaction of production system and year was significant for wheat production. Reasons for

**Table 5.** Partial and total ecological footprints of wheat and spelt production for the years 2008–2010 (ha ha$^{-1}$) for 1 ha of production area.

| | Footprint category | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Production system** | **Production area** | **% of total footprint** | **Seeds** | **% of total footprint** | **Machinery use** | **% of total footprint** | **Fertilization and plant protection** | **% of total footprint** | **Total footprint** |
| **Wheat** | | | | | | | | | |
| CON | 1.00 | 1 | 3.98a | 5 | 6.88c | 8 | 73.36a | 86 | 85.21a |
| INT | 1.00 | 2 | 3.40a | 5 | 6.90c | 11 | 52.25b | 82 | 63.54b |
| ORG | 1.00 | 10 | 0.69b | 7 | 8.45b | 83 | 0c | 0 | 10.15c |
| BD | 1.00 | 9 | 0.58b | 5 | 9.03a | 85 | 0c | 0 | 10.61c |
| Control | 1.00 | 14 | 0.46b | 6 | 5.63d | 79 | 0c | 0 | 7.09c |
| **Spelt** | | | | | | | | | |
| CON | 1.00 | 2 | 4.90a | 8 | 7.05b | 11 | 48.65a | 79 | 61.60a |
| INT | 1.00 | 2 | 3.60b | 8 | 7.01b | 15 | 33.64b | 74 | 45.24b |
| ORG | 1.00 | 10 | 0.85c | 8 | 8.45a | 82 | 0c | 0 | 10.30c |
| BD | 1.00 | 9 | 0.77c | 7 | 9.25a | 84 | 0c | 0 | 11.02c |
| Control | 1.00 | 14 | 0.62c | 9 | 5.63c | 78 | 0c | 0 | 7.25c |

Mean values of 3 years are presented. Different letters indicate statistically significant differences between production systems at 95% probability (Duncan test).

**Table 6.** Overall footprint per unit ($a_{tot}$), SPI and EEP for wheat and spelt production depending on production system and year.

| | Wheat | | | Spelt | | |
|---|---|---|---|---|---|---|
| **Factor** | **$a_{tot}$ (m$^2$ kg$^{-1}$)** | **SPI** | **EEP (kg ha$^{-1}$)** | **$a_{tot}$ (m$^2$ kg$^{-1}$)** | **SPI** | **EEP (kg ha$^{-1}$)** |
| **Production system (PS)** | | | | | | |
| Control | 31 ± 3b | 0.48 ± 0.05b | 349 ± 30a | 43 ± 3c | 0.65 ± 0.04c | 246 ± 15a |
| CON | 230 ± 22a | 3.33 ± 0.28a | 49 ± 4c | 280 ± 19a | 4.26 ± 0.29a | 37 ± 2c |
| INT | 204 ± 19a | 3.00 ± 0.30a | 58 ± 7c | 207 ± 21b | 3.16 ± 0.32b | 53 ± 5c |
| ORG | 47 ± 5b | 0.72 ± 0.08b | 242 ± 26b | 52 ± 4c | 0.79 ± 0.06c | 202 ± 12b |
| BD | 37 ± 3b | 0.57 ± 0.05b | 296 ± 29ab | 49 ± 4c | 0.75 ± 0.06c | 219 ± 21ab |
| **Year (yr)** | | | | | | |
| 2008 | 137 ± 28a | 1.99 ± 0.39a | 165 ± 26 | 125 ± 23 | 1.90 ± 0.35 | 155 ± 23 |
| 2009 | 96 ± 19b | 1.39 ± 0.26b | 215 ± 37 | 137 ± 29 | 2.08 ± 0.45 | 154 ± 23 |
| 2010 | 97 ± 19b | 1.48 ± 0.29b | 217 ± 34 | 118 ± 20 | 1.79 ± 0.31 | 145 ± 22 |
| **ANOVA** | | | | | | |
| PS | *** | *** | *** | *** | *** | *** |
| Y | *** | ** | n.s. | n.s. | n.s. | n.s. |
| PS × Y | ** | * | n.s. | n.s. | n.s. | n.s. |

Means ± SEs are presented. Different letters indicate statistically significant differences for each factor and indicator separately at 95% probability (Duncan test). Levels of significance: n.s. – non significant ($P>0.05$); *$P \le 0.05$; **$P \le 0.01$; ***$P \le 0.001$.

these differences between years can be found in lower average yields in the year 2008, where inputs into the systems remained on a similar scale as in the following 2 years.

Ratios between farming systems, where control = 1, provide us with a visual overview as to what influence production systems as such and yields have on the performance of farming systems under study (Fig. 1). Results indicate that higher yields in CON and INT systems can partly compensate for their high footprints. However, there still remains a 6 : 1 ratio for the $a_{tot}$ and SPI between the CON and control system, where EEP does not rise above the ratio 0.2 : 1 for CON : control, whereas it is 0.7–0.9 : 1 for the ORG/BD : control systems, respectively.

Currently, the ORG : CON farmed land ratios in the EU lie from 1 : 830 (Malta), to 1 : 15.4 (Slovenia) and up to 1 : 6.5 (Austria), with the EU-27 average amounting to 3.9% of the total agricultural area being managed organically[22]. Where does that leave us in the future, when we take into account the results from this trial? One of the main objectives to organic farming is that it does not produce enough food to feed the whole population—now and in the future[23]. However, several research projects and reports have demonstrated otherwise[24,25]. Even if yields in

**Table 7.** Land use for wheat and spelt production in Slovenia in 2010 with the corresponding ecological footprint and projected change with the eventual change of farming practice in 2015 and 2050.

| | Wheat | | | | Spelt | | | |
|---|---|---|---|---|---|---|---|---|
| **Production system (PS)** | **Area (ha)** | **% of total wheat area** | **Resulting footprint (ha)** | **Total yield (kg)** | **Area (ha)** | **% of total spelt area** | **Resulting footprint (ha)** | **Total yield (kg)** |
| **Situation in 2010**[1] | | | | | | | | |
| CON | 18,599 | 59 | 1,584,913.60 | 79,288,006 | 50 | 23 | 3081.81 | 113,068 |
| INT | 12,885 | 41 | 818,688.16 | 47,456,081 | 24 | 11 | 1090.84 | 57,117 |
| ORG | 206 | 1 | 2081.29 | 502,544 | 143 | 66 | 1472.86 | 291,536 |
| BD | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Total | 31,690 | 100 | 2,405,683.05 | 127,246,631 | 217 | 100 | 5645.51 | 461,721 |
| **Projection 2015**[2] | | | | | | | | |
| CON | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| INT | 25,352 | 80 | 1,610,796.14 | 93,371,416 | 282 | 10 | 12,738.14 | 666,971 |
| ORG | 3552 | 19 | 36,041.01 | 8,702,400 | 2252 | 80 | 23,201.56 | 4,592,497 |
| BD | 187 | 1 | 1984.60 | 595,782 | 282 | 10 | 3102.83 | 686,960 |
| Total | 29,091 | 100 | 1,648,821.75 | 102,669,598 | 2815 | 100 | 39,042.53 | 5,946,427 |
| **Projection 2050**[3] | | | | | | | | |
| CON | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| INT | 3169 | 10 | 201,349.52 | 11,671,427 | 0 | 0 | 0 | 0 |
| ORG | 14,625 | 70 | 148,395.19 | 35,831,250 | 7948 | 80 | 81,868.66 | 16,205,010 |
| BD | 4178 | 20 | 44,340.35 | 13,311,108 | 1987 | 20 | 21,897.17 | 4,847,992 |
| Total | 21,972 | 100 | 394,085.07 | 60,813,785 | 9934 | 100 | 103,765.83 | 21,053,002 |

[1] Data for area obtained from Ministry of Agriculture, Forestry and Food of Slovenia on the basis of an official email inquiry, based on the number of farms that receive subsidies. Data for yields per hectare are taken from this trial, where organic wheat yields were lower than in previous studies. This is also the reason for the greater reduction in total yields in the 2015 and 2050 projection.

[2] According to the Slovene Action Plan for Organic Farming[35]), 20% of the utilizable agricultural area is planned to be converted to organic farming in 2015. Moreover, due to the planned change of the Common Agricultural Policy and the Slovene Agri-Environmental Programme after 2013, only integrated and organic (biodynamic) farming is under consideration to be subsidized in the future. The proportion of spelt in the total area is projected to increase with years, as spelt is more resistant to pests and diseases and is relatively undemanding and less susceptible to fluctuations in growing conditions. Also less fertilizer input is demanded.

[3] Projection is based on the assumption that by 2050, conventional energy sources (oil, gas, coal and nuclear power) will be in decline and also expensive, thus a shift toward more sustainable, ecologically intensive and less energy-intensive agricultural systems will be a matter of need, rather than choice.

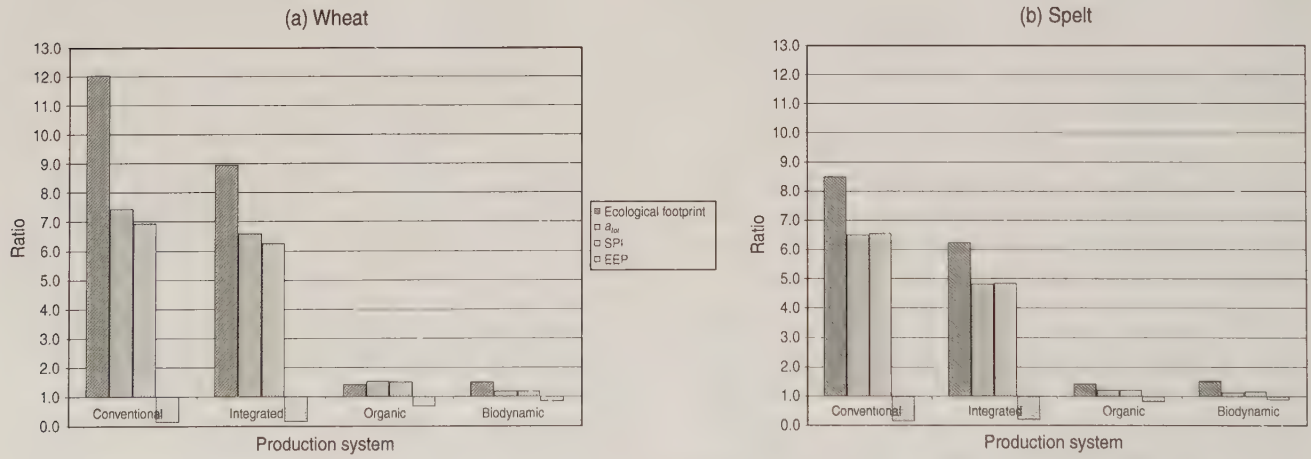


**Figure 1.** Ecological footprint, overall footprint per unit ($a_{tot}$), SPI and EEP ratios (control = 1) between production systems for 3 years of wheat (a) and spelt (b) production.

developed European countries, where CON industrial agriculture is now predominant, would be 5–20% lower with ORG and BD agriculture, population projections for developed countries in the next 50 years partly coincide with these lower yields[26]. Next to that, a large proportion of the currently produced grain goes toward feeding animals. In Slovenia alone, feedstuffs for animals are produced on more than three-quarters of the arable land[21]. High

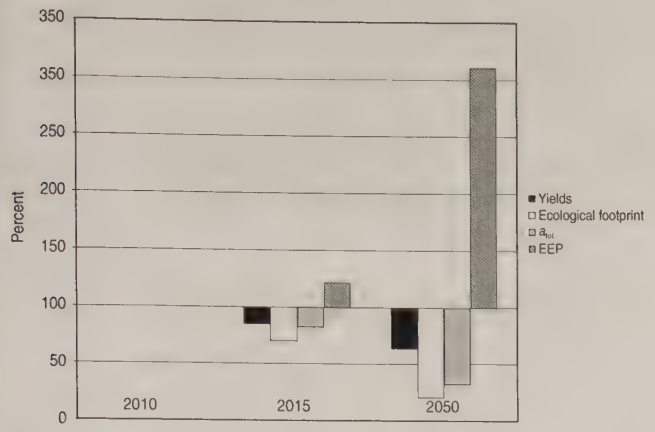


**Figure 2.** Projected change of total yields, ecological footprint, overall footprint per unit ($a_{tot}$), and EEP from 2010 ( = 100%) to 2050 for wheat and spelt production in Slovenia.

competition for grain and animal manures is also present in the developing 'bio-gas' sector[27]. In this sense there is a relatively great reserve in arable land, which could be used for food, instead of feed or energy production. Taking it a step farther from food production levels, what will happen when oil reserves get depleted? It is important to keep in mind that the relation between population and oil extraction is one of cause and effect. With greater use of mineral fertilizers and pesticides, production of which is based on conventional energy sources, higher yields were achieved in the past century and therefore more people could be fed from the same area than before. However, the downside of this advance in agricultural production is also visible in the results of this research—the high proportion of the final footprint going to mineral fertilizer and pesticide production, caused by using those conventional energy sources and therefore needing a large area to offset the high environmental burden. With abundant oil, a large population is possible—ignoring, of course, the fact that environmental degradation may diminish the human population. Without abundant oil, on the other hand, a large population will not be possible[28]. Fossil oil renewal rates and quantities are much slower than the current usage rates and quantities[28–30], and eventually the era of cheap fossil oil will come to an end. With this in mind we projected the magnitude of change, if all land for wheat and spelt production in Slovenia were converted to organic/biodynamic farming in 2050 (Table 7). Production levels would be lower by almost a third, the ecological footprint and $a_{tot}$, however, would be lower by almost two-thirds (Fig. 2). Consequently, the EEP would rise threefold compared to the current situation. In 2009, around 170,000 tons of wheat were consumed by the Slovenian population, and an additional 100,000 tons were used for animal feed. More than 45% of that wheat had to be imported[21]. This means that only to nourish people (in order to be self-sufficient) in 2050, twice as much arable land would have to be devoted to wheat production, assuming, of course, the same production levels as with current production techniques. But how can we tackle this issue in the future? Possible solutions can be sought in changing crop rotations and/or land use (as mentioned previously, more than 80,000 ha of land is currently devoted to maize for grain or silage production—mainly for animal feed), but foremost also improving and further developing current alternative agricultural production practices and techniques. In addition, Ewert et al.[31] argue that production levels of the main crops in Europe will rise in the following decades (owing to improved production techniques and a changed climate), and thus less land will have to be cropped to produce the same amount of food. As mentioned previously, efficient use of machinery and the invention of new forms of working the soil will be of crucial importance. Some good examples pointing toward the future can already be seen in practice. One of them is the Eco-Dyn System, where fuel use per hectare has been lowered to 20–30 litres $ha^{-1}$ (it amounted to more than 90 litres $ha^{-1}$ in our study); yields, however, remained stable around the average yields of Germany[32]. Another example is the reinvention and improvement of the ridge-till system[33] in order to lower machinery use and improve the quality of soils and consequently the health of plants and quantity of produce. In both cases the farmers are, next to the technological innovations, using a biodynamic approach toward farm and soil management, which has been found to improve soil fertility in several studies[34]. However, all these improvements and approaches need further research and development in order to adapt them to different microclimatic, pedologic and cultural conditions.

## Conclusions

Our results indicate critical points in production of wheat and spelt in each production system, where greatest improvements could be achieved by abandoning mineral fertilizer and pesticide use in industrial farming systems. However, machinery use also needs attention in the near and distant future in all systems studied in order to improve all the recorded parameters and to get closer to sustainable farming systems from a productive and environmental point of view. All mentioned changes will, of course, have to be thought through carefully and support by government policies, economic incentives as well as grassroots activities will aid in making them successful.

The question that stakeholders in agriculture will have to ask in the following years is: Can we save and/or produce enough resources for the current and future generations, when we use or leave an impact on almost 80 ha of surface area to produce 1 ha of wheat (or in a similar size range any other crop)? Or do we have to rethink and above all change the way we farm, live and make decisions in order to survive on planet Earth?

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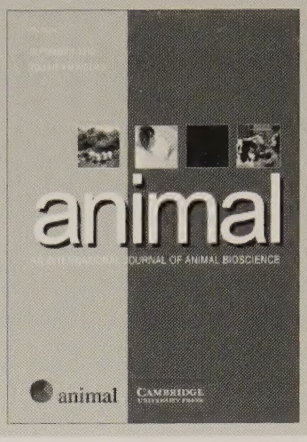


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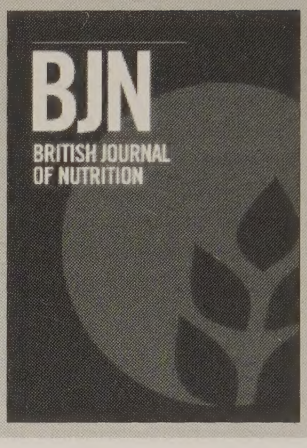


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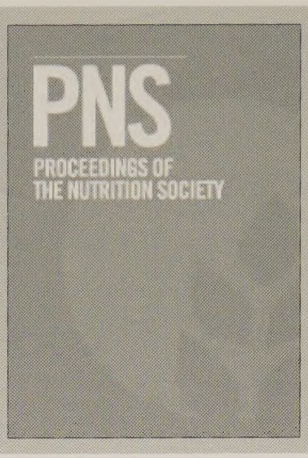


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*Kathleen Hilimire* 173

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*Matt Liebman, Rhonda L. Graef, Daniel Nettleton, and Cynthia A. Cambardella* 180

A capacity assessment of New England's large animal slaughter facilities as relative to meat production for the regional food system
*Chelsea Bardot Lewis, and Christian J. Peters* 192

Soil fertility and crop yields in long-term organic and conventional cropping systems in Eastern Nebraska
*Sam E. Wortman, Tomie D. Galusha, Stephen C. Mason, and Charles A. Francis* 200

Using reference values to assess environmental sustainability of dairy farms
*I. Acosta-Alba, M.S. Corson, H.M.G. van der Werf, and P. Leterme* 217

Technical efficiency of organic pasture farming in Germany: The role of location economics and of specific knowledge
*S. Lakner, S. von Cramon-Taubadel, and B. Brümmer* 228

Ecological impact of wheat and spelt production under industrial and alternative farming systems
*Martina Bavec, Michael Narodoslawsky, Franc Bavec, and Matjaž Turinek* 242

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